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BINBROOK RESERVOIR (GLANBROOK TOWNSHIP) WATER QUALITY ASSESSMENT AND MANAGEMENT IMPLICATIONS

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BINBROOK RESERVOIR (GLANBROOK TOWNSHIP) WATER QUALITY

ASSESSMENT AND MANAGEMENT IMPLICATIONS

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PREFACE

Binbrook Reservoir was one of 12 southwestern Ontario lakes investigated by the Ontario Ministry of Environment and Energy (MOEE) as part of the Environmental Monitoring and Reporting Branch, Inland Lakes Program. The program was designed to monitor a selection of hardwater inland lakes with, or with the potential for, nuisance blue-green algae blooms and other water quality problems. Lakes were chosen in response to complaints regarding surface scums, algal blooms and seasonal hypolimnetic anoxia, received from organized public associations, MOEE regions and/or Conservation Authorities. Based on these surveys, several lakes were assessed as having a good potential to respond to experimental treatment programs.

Background limnological data were collected on Binbrook Reservoir during 1988 and 1989. This report was prepared to ensure that the water quality information was made available to those agencies and individuals expressing an interest in the findings, and to provide the technical basis for possible future water quality management initiatives.

ABSTRACT

Binbrook Reservoir is a hardwater, polymictic, eutrophic reservoir experiencing mean annual water clarity of less than 1.0 m. Total chlorophyll a concentrations that were weakly correlated to Secchi disc visibilities and the absence of a significant chlorophyll a/turbidity relationship, combined with a negative correlation between Secchi disc and turbidity suggested that water clarity was principally governed by non-algal turbidity. Approximately 63% of the variation in Secchi depth was explained by:

$$\ln \text{Secchi disc(m)} = -0.029 \text{ turbidity (FTU)} - 0.15$$

Turbidity levels were highest in the upper regions of the reservoir, as were TP concentrations. Total phosphorus was also highly correlated with turbidity levels:

$$\text{Total phosphorus(mg/L)} = 0.013 \text{ turbidity(FTU)} + 0.0247 \; (r=0.813),$$

suggesting that phosphorus loads were runoff related and a result of agricultural practices in the Welland River watershed.

The reservoir experienced high spring nitrate levels with a linear seasonal nitrate depletion rate in both years of the study.

Periodically during stratification, the hypolimnion experienced anoxia. During these periods increases in hypolimnetic phosphorus, ammonium and manganese levels were observed, suggesting that sediment nutrient release was occurring. Euphotic zone TN:TP ratios peaked during mid-summer and fell through the second half of the season coinciding with reports of nitrogen fixing blue-green algal blooms in the late summer of each year.

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INTRODUCTION

Binbrook Reservoir, Glanbrook Township (43°6′ / 79°50′) was formerly known as Lake Niapenco (Figure 1). It was constructed in 1971 with the primary purpose of a storage reservoir providing downstream flood protection and augmenting low summer flows. Summer outflows were maintained at a constant minimum volume of 5 cfs (140 Litres/second). Outflow was near-bottom draw (2 metres off bottom), through a sixteen inch diameter pipe with electronically controlled valves, governed by reservoir head, through pressure sensitive transducers. Dam flow and drawdown specifics, rule curves and operational information during flood conditions are summarized in NPCA (1987). The Welland River is the primary inflow. Levels are lowered by approximately 4 m in the autumn of each year, to provide for storage of the following year's spring freshet. During the spring, lake levels rise to the design storage capacity of 5.551×10^6 m^3. As a multi-purpose reservoir it is also intended to furnish a base for a water oriented conservation area and for the promotion of wildlife, recreation and education (NPCA 1978), and is operated by the Niagara Peninsula Conservation Authority (NPCA).

The watershed (4,140 hectares) is relatively flat with low morainic ridges of relatively impervious clay till. Land use is primarily agricultural with large beef and dairy operations but the production of grains and hay is also popular. Some strip development occurs along major roads. 70% of household septic systems within the watershed failed to comply with regulations (Laidley 1991). Grey water bypassing was the primary reason for non-compliance. Additional watershed details and descriptions are available in Laidley (1988) and Laidley (1991).

The reservoir is divided into two segments by a weir at the western end of the lake which was built in 1981. The weir, constructed of earth fill, protected with rip-rap and cement, uses a gate valve controlling water flow to create a holding pond in the upper one-third of the reservoir. It contains an estimated 5.98×10^5 m^3 of water (1981 estimate) and was designed to provide a constant wetland area for staging waterfowl. This area remains

flooded throughout the year independant of main reservoir water levels. In its current design the upper wetland may also act as a partially effective sedimentation basin providing a limited form of water quality protection to the reservoir because of wetland nutrient uptake and the sedimentation characteristics of a holding basin (OMOE 1991). The lower reservoir contains virtually no macrophyte beds. The reservoir is surrounded by a 400 hectare park. Key reservoir physical data are provided in Table 1.

Trap net catches in 1992 conducted by the Ontario Ministry of Natural Resources (MNR, Bob Lewies, pers. comm.) resulted in a total of 14,181 fish caught of which 13,511 were crappie. Fish community estimates based on these numbers showed that the lake was dominated by Black Crappie (*Pomoxis nigromaculatus*) (95%), with lesser numbers of Carp (*Cyprinus carpio*) and Brown Bullhead (*Ictalurus nebulosus*) (1 to 3%). Higher levels of predatory sport fish such as Northern Pike (*Esox lucius*), Walleye (*Stizostedion vitreum*) and bass (*Micropterus* spp.) also inhabit the reservoir but were estimated to comprise less than 1% of the total fish population. Beneficial uses of the lake apart from sport fishing include an active autumn waterfowl hunting season, windsurfing and swimming.

Lake specific problems include turbid water resulting in poor water clarity, high nutrient loadings, hypolimnetic anoxia, blue-green algae forming nuisance blooms in the late summer, and beach closures due to high bacterial levels. Non-point source nutrient loadings and bacterial problems are currently being addressed under the auspices of the Binbrook Reservoir, Clean Up Rural Beaches (CURB) plan (Laidley 1991). This plan involves the principal remedial option of an ecosystem approach to sustainability through a rural land stewardship concept, thus managing the reservoir by attempting to control anthropogenic nutrient and bacterial sources. Public information programs, encouraging adoption of environmentally sound agricultural practices and the dissemination of information on proper land management are important components of the plan (Laidley 1988).

Declines in public use through the mid-1980s resulted in a series of measures initiated by the NPCA to try and revive public interest in the park. These included the construction and

installation of a chlorinated beach enclosure for swimming. During the first decade of operation, reservoir attendance regularly exceeded 70,000 visitors per summer (currently between 10,000 and 20,000). Aesthetic and health concerns because of elevated bacterial levels commencing in 1983, and blue-green algae blooms, were considered the main causes of the declines in attendance. This report provides a detailed water quality summary and assessment of the main reservoir and summarizes several potential management options.

METHODS

Water samples, biological samples and oxygen/temperature profiles were collected approximately bi-weekly between late April and mid-September during 1988 and 1989. Samples were collected from three sites (Figure 1); approximately 30 m from the morning glory spillway at the far east end of the lake (BR1), near the center of the lake at Trinity Church Road (BR2), and from moving water at the inflow to the reservoir, the Welland River at Tyneside Road (BR3).

Water clarity was determined with a Secchi disc. The Secchi disc is a 20 cm diameter steel plate suspended from a marked rope and is painted in alternating black and white quarters. The disc was lowered on the shaded side of the boat until it disappeared, then raised again until it reappeared. The average of these two depths was termed the Secchi disc depth.

Two weighted, narrow mouthed (2.5 cm) 1-litre glass bottles were lowered and raised through the euphotic zone (twice the Secchi disc depth), collecting a composite sample of the water column. The rate of descent and ascent was timed so that the bottles were completely filled just as they reached the surface. (Note: The calculated euphotic zone was limited to twice the Secchi disc depth to a maximum of 1 metre off bottom, so that at no time did euphotic zone samples include water from below 1 metre off bottom).

A 500 mL subsample was withdrawn into a separate bottle for chlorophyll analysis, which was stabilized with 2 mL of a 2% $MgCO_3$ suspension. A second 500 mL subsample for

metals analyses was preserved with 2 mL of concentrated nitric acid to prevent metals precipitation. A third subsample was preserved with Lugol's iodine for phytoplankton analyses and a fourth sample for nutrient chemistry was unpreserved. Samples were kept cool and delivered to the MOEE's laboratory the same day for analyses. Samples were refrigerated and generally analyzed within 48 hours of collection following standard procedures (OMOE 1981). A second set of samples was drawn from 1 meter off the bottom of the lake (1 MOB), using a 6 litre capacity PVC Van-Dorn bottle. 500 mL samples were withdrawn from the bottle for nutrient chemistry and metals analyses. Phytoplankton analyses were conducted on a recombined basis (the bi-weekly samples were pooled into a single representative April to October sample). Phytoplankton taxa were identified to the genus level according to Nicholls et al. (1977).

At station BR1 and BR2, oxygen and temperature measurements were taken at one metre intervals from the surface of the lake to the bottom, using a YSI 58 dissolved oxygen and temperature metre. The metre was calibrated against a standard hand held mercury thermometer for temperature accuracy, and calibrated against water samples of known oxygen concentration, determined by the micro-Azide modification of the Winkler technique. In addition, 60 mL Winkler samples from the lake surface and 1 MOB were taken to confirm the integrity of oxygen metre readings at these depths.

At station BR3 (Welland River inflow) samples were taken as grab samples from moving water. From this site, samples were collected for nutrient chemistry and metals analyses only.

Zooplankton samples were collected using a modified, metered Clarke-Bumpus net, with 80 μm mesh and a 15.5 cm mouth. The collection net was raised through a vertical column from 1 MOB to the surface with the net attached to collect the sample (A), and with the net removed (B), to determine net flow restriction. Efficiency was determined by the ratio A/B from the meter values of each haul. Net efficiency was used to adjust biomass and density counts. Zooplankton collections were analyzed according to Yan and Mackie (1987).

Biomass values are presented as dry weight. Sample enumeration was conducted as described in Girard and Reid (1990).

Periods of high and low runoff were estimated from the nearest Environment Canada water levels gauge on the lower Welland River in an adjacent watershed (Environment Canada). Stream gauging and levels monitoring were not however conducted on the inflow waters to Binbrook Reservoir.

Statistics

All correlations are Pearson product-moment, and significant at $\alpha=0.01$ unless otherwise indicated. T-tests are significant at the $\alpha=0.01$ level unless otherwise indicated. Stepwise linear regression was performed in a forward manner. Wilkinson (1990) was the computer software used for the majority of the statistical analyses.

RESULTS

Appendix A through Appendix E list data summaries by station and year and can be referred to for parameter specific means, standard deviations, minima and maxima. Appendix F lists selected raw water quality data for the euphotic zone stations and the Welland River inflow.

Oxygen and Temperature

Figures 2 to 5 graphically describe oxygen and temperature trends in Binbrook Reservoir during 1988 and 1989. At both stations temperatures reached their seasonal maximum in late July of the year. Maximum reservoir surface temperatures recorded, reached 26° C and were generally higher (usually by 1° C) at Station BR1 compared to station BR2. Stratified layers formed at the deeper station BR1 by late June in both years sampled, with a distinct hypolimnion visible by mid-July. Anoxic conditions were observed below 5.5 m throughout

July and August in 1988 and 1989. Oxygen levels below 4.5 m at station BR1 were generally unfavourable to fish by mid-summer in both years with levels less than 5.0 mg/L. Oxygen concentrations were higher in the bottom waters of station BR2, not falling below 5.0 mg/L in 1988, except during July of 1989 when oxygen levels temporarily fell to less than 2.0 mg/L. Wind mixing at this shallow station generally maintained destratified conditions.

Phosphorus

Soluble reactive phosphorus concentrations [SRP] peaked during the early season reaching levels as high as 32 μg/L at BR2 but declined as the season progressed (Figure 6). Euphotic zone and 1 MOB [SRP] were highly correlated ($r=0.830$) at both sites except during the late summer of 1989, when at station BR1 bottom water concentrations exceeded euphotic zone levels (T-test, $\alpha=0.05$), suggesting sediment nutrient release. Mean annual [SRP] at BR2 were approximately twice levels observed at BR1, ranging between 9 to 14 μg/L and rarely fell below the detection limit. Increased autumn [SRP] at 1 MOB and in the euphotic zone were common at Station BR2 but not typical at Station BR1.

Seasonal total phosphorus concentrations [TP] displayed trends similar to the [SRP] trends at both stations (Figure 7) with the exception of a fall hypolimnetic increase at station BR1, which did not occur. T-tests showed no significant difference between euphotic zone and 1 MOB concentrations at BR1 but significantly higher 1 MOB levels were observed during both years at station BR2. Euphotic zone annual means were higher at BR2 averaging 65 μg/L compared to 43 μg/L at BR1.

[TP] and turbidity were significantly correlated (Table 2) in both the euphotic zone ($r=0.813$), and at 1 MOB ($r=.947$). A linear relationship combining 1 MOB and euphotic zone values (Figure 8a) allowed the prediction of [TP] from turbidity according to the model:

$$\textbf{[TP] (mg/L)} = 0.013 \textbf{ (turbidity)} + 0.0247$$

$$n=88,\ r=.890,\ \alpha=0.01$$

Reservoir levels of [TP] and [SRP] were also significantly correlated (r=0.885).

Phosphorus concentrations from the inflow (BR3) displayed mid-summer seasonal peaks surrounded by spring and fall minima (Figure 9). Annual inflow means however (272 and 196 μg/L), were three to four times greater than reservoir levels.

Secchi Disc, Turbidity and Chlorophyll a

Mean annual Secchi disc visibility ranged between 0.39 m at station BR2 in 1988 to 0.64 m at station BR1 during 1989. Maximum visibility (1.7 m) occurred in August of 1989 at BR1 (Figure 10). A clear water phase was observed through July and early August of 1989 at station BR2.

Chlorophyll a concentrations displayed early summer minima and fall peaks at station BR1. At BR2 a late summer chlorophyll increase was evident by early July (Figure 10). Annual means were significantly higher in 1988 and reached 15.9 μg/L at BR2 and 12.3 μg/L at BR1.

A significant negative correlation between total chlorophyll a and TN:TP ratios (r=-0.591) occurred, but the expected relationships between chlorophyll a and [TP], [SRP] or Secchi disc were absent (Table 2).

Reservoir turbidity was consistently higher in the early season and station BR2 was generally more turbid compared to station BR1 (Figure 11). While no significant difference between euphotic zone and bottom turbidity was measured at station BR1 (1989 annual mean, 15.3 FTU), bottom waters at station BR2 experienced consistently higher turbidity compared to euphotic zone levels. An average annual euphotic zone mean of 40 FTU was measured.

The correlation between Secchi disc and total chlorophyll a was weak but significant (r=-0.380, α=0.05). There was no significant relationship between turbidity and

chlorophyll a (Table 2). A strong correlation between Secchi disc visibility and turbidity was apparent (r=-0.672). This combination of relationships suggests that water clarity fluctuations were principally a result of non-algal turbidity.

An empirical relationship between Secchi disc and chlorophyll a was modelled to predict changes in transparency that could be expected from lower chlorophyll levels, but correlation was weak (Figure 8b). Only 17% of the Secchi depth variation was explained, and if the only extreme chlorophyll value was deleted, the regression was not significant. Therefore, water clarity relationships between turbidity levels and [TP] in Binbrook Reservoir were examined (Figure 8c and 8d) and explained 63% and 36% respectively, of the variation in Secchi depth according to:

a) $\boldsymbol{\ln \text{Secchi disc(m)} = -0.029 \text{ (turbidity)} - 0.15}$

$\boldsymbol{n=44,\ r=0.793,\ \alpha=0.01}$

b) $\boldsymbol{\ln \text{Secchi disc(m)} = -11.94 \text{ (TP mg/L)} - 0.102}$

$\boldsymbol{n=45,\ r=0.596,\ \alpha=0.01.}$

Forward stepwise linear regression from turbidity, chlorophyll a, [TP], and nitrogen levels resulted in a significant bivariate model. The equation explained just over 60% of the variation in Secchi disc measurements:

$$\boldsymbol{\ln \text{Secchi disc (m)} = 0.005 - 0.167 \ln \text{(Turb.)} - 0.055 \ln \text{(Chlorophyll } \underline{a}\ \mu\text{g/L)}}$$

$$\boldsymbol{R^2=.620,\ MSE=0.407,\ F=30.951,\ \alpha=0.001}$$

Introducing chlorophyll a into the model, however, weakened the relationship ($R^2=.620$ compared to $R^2=.630$ for eq'n a), further supporting the premise that non-algal turbidity was the principal factor determining water clarity in Binbrook Reservoir.

At station BR3 (Welland River inflow) turbidity levels were much higher than reservoir

levels and fluctuated widely through all seasons (Std dev. $\pm$ 39). Annual means between 78 and 98.9 FTU were measured. Turbidity that peaked at 200 FTU following a summer storm (Figure 11), was observed in 1989. During stagnant, low flow conditions, turbidity in the Welland River generally improved with levels as low as 2.4 FTU.

Nitrogen

Total nitrate concentrations were similar between sites and depths within years, but were higher in 1989 compared to 1988 levels. Spring maxima in 1989 were 1.8 mg/L at station BR2 and 1.4 mg/L at station BR1. Nitrate depletion rates in the euphotic zone and at 1 MOB were virtually identical in both years averaging 0.009 mg/L/day (Figure 12). Minimum nitrate levels neared the detection limit with levels as low as 0.04 mg/L by mid-August in 1988, but higher levels were present in the late summer of 1989 ranging between 0.15 and 0.20 mg/L.

Ammonium concentrations at both lake stations were significantly higher at 1 MOB compared to the euphotic zone (Figure 13). Mean annual 1 MOB concentrations were highest at BR1 during 1989 (0.170 mg/L). These were a result of late summer increases (Max. 0.384 mg/L), which occurred during periods of stratification and anoxic hypolimnia. Increases in euphotic zone levels were concurrent, but were less than 50% of 1 MOB concentrations. Distinct but weak thermal stratification (Figures 2 and 4) at this time implied that entrainment of nutrient rich bottom waters into surface waters was plausible explanation for the increases in the euphotic zone.

TN:TP ratios (Figure 14) varied seasonally across the reservoir reaching peak levels during mid-July of both years. TN:TP ratios fell to near or below 20:1 during late August of 1988 at both reservoir stations and at BR2 in 1989. Ratios were consistently higher at BR1 reaching 64:1 during 1989, with an annual mean of 45:1, compared to 26:1 at BR2.

Nitrogen levels from the inflow samples (Figure 15) did not show a consistent seasonal

pattern. Mid-summer nitrogen peaks were measured in 1988, but in 1989 concentration peaks occurred twice, once in late May and again in late July. A review of concurrent reservoir levels (D. Watson (NPCA), pers. comm.) and Welland River stream gauge measurements (Environment Canada) from the adjacent watershed indicated that higher flows/runoff likely occurred during these time frames. Increased flows and higher concentrations, a result of surface runoff from fertilized fields in the Welland River watershed may in part have combined with autochthonous sources to raise euphotic zone levels within the reservoir during the late season of 1989.

Conductivity and Chloride

Reservoir conductivity was higher in 1989 compared to 1988. Annual means in 1989 (567 to 565 μmhos/cm) were approximately 80 μmhos/cm greater than in 1988. At both reservoir stations no significant difference (t-test, $\alpha = 0.0001$) between conductance at 1 MOB and the euphotic zone was measured. Levels within years were slightly higher however at BR2 compared to BR1 particularly during the early season (Figure 16). This indicated the greater impact of high spring runoff volumes on water quality of the upstream portions of the reservoir. A trend to lower conductance was observed in the late summer of each year in response to the diluting effect of increased reservoir levels and sharp declines in the conductivity of the Welland River (Figure 16).

Mean annual chloride concentrations were higher in 1989 compared to 1988 increasing to 62.8 mg/L at station BR2 and 61.8 mg/L at station BR1. 1988 mean levels were 45.6 mg/L and 46.9 mg/L, respectively. Chloride concentrations were highly correlated to conductivity ($r = 0.981$) as were iron, manganese, magnesium, potassium and sodium levels (Table 2). These ions were used to develop an equation predicting reservoir conductivity from the dominant ions chloride and magnesium, using forward stepwise linear regression.

$$\text{Conductivity } (\mu\text{mhos/cm}) = 176.887 + 4.375 \text{ Cl(mg/L)} + 7.926 \text{ Mg(mg/L)} \quad R^2=0.989,$$
$$MSE=39029.36, F=901.393, \alpha=0.001$$

Iron and Manganese

Iron levels [Fe] were slightly higher at station BR2 compared to BR1, particularly during the spring. The seasonal decline from spring peaks was gradual (Figure 17). Higher 1 MOB [Fe] during the late season, expected in response to anoxic bottom water levels, was not observed. Annual mean concentrations varied between 0.473 mg/L at station BR1 in 1989 to 1.362 mg/L at station BR2 in 1988.

Manganese [Mn] concentrations, however, did increase as a result of hypolimnetic anoxia or even low oxygen (<2.0 mg/L) conditions at both station BR2 and station BR1 (Figure 18). During even short periods of low oxygen concentrations (Figures 2 to 5), 1 MOB [Mn] exceeded euphotic zone levels clearly suggesting a drop in sediment redox levels and that other nutrient release was imminent. Recovery to lower [Mn] was observed following wind induced mixing, or fall turnover in late September. Seasonal Welland River trends are presented in Figure 19.

Calcium, Alkalinity, Hardness and pH

Seasonal euphotic zone trends are presented in Figure 20. No seasonally significant trends were observed except for pH which displayed a typical mid-summer rise normally associated with increased primary productivity during this season. Mean annual hardness levels were slightly higher in 1989 compared to 1988 described by an annual mean of 208 mg/L at station BR1.

Phytoplankton

Thirty-six genera, typical of eutrophic hardwater reservoirs were collected from euphotic zone samples during 1988 and 1989 (Table 3). During 1988 the dominant blue-green algae during late-summer and fall blooms were the nitrogen-fixing *Aphanizomenon* and *Anabaena*. At this time TN:TP ratios were below 20:1. In 1989 *Anabaena* remained dominant at station BR2, but *Lyngbya* was the only blue-green algae present in samples collected from BR1. Mean annual total biovolumes ranged between 0.630 mm^3/L at station BR1 to 4.338 mm^3/L at station BR2 (Figure 21).

Blue-green algae (Cyanophytes) developed nuisance late summer blooms in 1988 at both stations, BR1 and BR2, and comprised over 23% of the mean annual recombined algal assemblage. Blue-green dominance fell to 6% at station BR2 and 0.2% at BR1 in 1989. Nuisance blooms were not observed during site sampling in 1989, although reports of shoreline accumulations were made that year (Laidley, pers. comm.).

At station BR1 *Cryptomonas* was dominant, comprising up to 67% of the algal biovolume in 1989 and replaced the blue-green algae of 1988. Representation by the Chrysophytes primarily *Dinobryon* and *Mallomonas*, was also significant at 15%.

At station BR2, blue-green algal levels were lower in 1989 at only 6% of the algal community, compared to 23% in 1988. Cryptophytes clearly dominated the assemblage at BR2 with a peak of 75% (3.267 mm^3/L) in 1989. This probably represented a massive *Cryptomonas* increase likely to have occurred during mid-July. Peak chlorophyll levels without a corresponding decline in water transparency suggest the *Cryptomonas* bloom occurred at this time (Figure 10). Turbid reservoirs are often dominated by cryptomonads (Cuker et al. 1990). High TN:TP ratios would likely have prevented the formation of the nitrogen fixing blue-green algae. Indications that the blue-green algae (6% of total biovolume) represented in the 1989 recombined sample were from a bloom which occurred

during the late season were low TN:TP ratios (near 20:1), high chlorophyll concentrations and a sharp decrease in Secchi disc visibility during late August (Figure 10).

Diatoms (Bacillariophytes) were not significant contributors to the algal assemblage comprising less than 5% of the algal community on an annual basis. Based on data from similar studies in southern Ontario, the diatom community was most likely to have peaked during the spring and or fall of the year (Gemza 1991, Vandermeulen and Gemza 1991). Reservoir silica levels, however, which fell to near detection limits during the summer of 1988 (Appendix F) indicate that diatom numbers were probably highest during the mid-summer.

Zooplankton

Twenty-two zooplankton species were recorded during 1988 and 1989 from station BR1 and BR2 (Table 4). Zooplankton were summarized into four categories: Daphnid cladocerans, Non-daphnid cladocerans, Cyclopoid and Calanoid copepods.

Daphnia spp. were well represented and dominated the zooplankton community through most of the year. Daphnid biomass comprised 50% or more of the total zooplankton biomass during June and July (Figures 20 and 21) but was reduced by the end of July in 1988 to less than 50 mg/m^3 (Figure 21). Daphnid biomass generally increased following the spring decline of non-daphnid cladocerans.

Non-daphnid cladoceran community composition was highest in the early May to mid-June period with reduced peaks in late summer (Figure 22) surrounding a mid-summer minimum when they comprised less than 5% of the biomass.

Calanoid copepods were more important during 1988, exhibiting a strong early season dominance at station BR1 (80%) and over 50% of the biomass at station BR2. In 1989 they

comprised less than 10% of the biomass except during a short period of time in early June of that year.

The predatory cyclopoid copepods were present throughout the year at both stations comprising up to 50% of the biomass, but were absent from the early part of 1988, when calanoid copepods were dominant. Cyclopoid biomass generally peaked with bosminid biomass. Correlation to non-daphnid cladoceran biomass was strong ($r=0.755$).

Bosmina longirostris populations typically peaked in the spring of the year with densities that exceeded 1 x 10^6 m^3 at station BR2 in 1988 (Figure 22). *Daphnia galeata mendotae* trends were less predictable. Peak levels in the early season and through June were observed in 1988, but mid to late summer peaks occurred in 1989. *Diaptomus siciloides* typically reached its population apex in mid to late June, while the dominant cyclopoid predator *Mesocyclops edax* did not reach its highest numbers until late summer and was even absent from collections during the first several collections of each year at station BR2. *Daphnia parvula* was absent from collections during 1988 and first appeared during the latter half of 1989.

Seasonal decreases in length were plotted for the dominant species (Figure 25). Consistent seasonal trends were observed for *D. siciloides*, and somewhat for *M. edax*, which was smaller in the late summer of each year. Declining lengths during the early summer of 1989 were observed for *Bosmina longirostris*. Close correlation with cyclopoid biomass (Table 2) and the presence of cyclopoids in the spring of 1989 suggest that cylopoid omnivory was probable, and that fish predation was likely a lesser influence. *Bosmina longirostris* is a well known prey of *M. edax*..

DISCUSSION

Mean annual [TP] and ammonium concentrations exceeding provincial guidelines; turbid waters and a seasonally anoxic hypolimnion were the principal conditions degrading the

beneficial uses of Binbrook Reservoir. Weak relationships between chlorophyll a and Secchi disc parameters, combined with significant correlations between turbidity and [TP], and Secchi disc, turbidity and [TP], indicate the importance of non-algal turbidity in governing Binbrook Reservoir water clarity, and indirectly, trophic state. The phenomenon is not unique and has been observed, principally in artificial reservoirs (Lind 1986). The problem of bacterial contamination limiting body-contact water recreation was described elsewhere (Laidley 1991).

Contributing factors to the late summer blue-green algal blooms (as reported by NPCA staff), appear to be declining TN:TP ratios (Smith 1986), sediment nutrient regeneration as a result of hypolimnetic anoxia (Syers et al. 1973, Moore et al. 1992) and high turbidity, which limited light penetration favouring surface bloom algae (Spencer and King 1987). The late season blue-green algal blooms and the resulting high levels of chlorophyll a likely resulted in the strong negative correlation between chlorophyll and TN:TP ratios. Seasonal nitrate reductions were strong and predictable. Although nitrate levels typically fall seasonally in response to reduced summer inputs (Vandermeulen and Gemza 1991, Gemza 1991), the nitrate depletion rate in Binbrook indicated that replenishment from inflowing waters was further reduced as a result of enhanced wetland nitrogen removal during the growing season (Rogers et al. 1991).

An anoxic hypolimnion was present during nutrient and metal concentration increases measured at 1 MOB. This indicated that the sediments could be a significant source of nutrients. The early signs of the onset of sediment nutrient release included regular periods of increased manganese concentrations which occur at a much higher redox level compared to those for iron, phosphorus and ammonium (Moore et al. 1991). So while [TP] increases were observed only during longer periods of stratification, the potential for a more frequent and prolonged release is present. Higher 1 MOB manganese concentrations were observed regularly during periods of stratification and anoxia.

While erosion control was not overlooked in the Binbrook CURB plan (Laidley 1991), it placed a "**low priority**" on controlling sediment erosion to the reservoir, probably because its primary mandate was to control bacterial contamination. Evidence from this report, however, clearly identified that water clarity, SRP concentrations and TP concentrations were dependent on turbidity levels. The principal sources of the turbidity appear to be nutrient rich (clay) particulates from agricultural runoff in the watershed. Phosphorus adsorption to soil particles and hence transport have been observed (Holtan et al. 1988) and may play an important role in Binbrook reservoir. Clay particles also serve as substrate and vectors for microbial transportation. As a result, measures to reduce turbidity and phosphorus will likely reduce bacterial levels as well (OMOE 1991). Laidley (1991) summarizes some of the measures used to protect rivers and streams, as applied to the Binbrook situation. These include improved manure storage and manure spreading techniques, vegetated buffer zones for riparian zone enhancement, woodlot management, and limiting cattle access to river-beds through improved fencing, as well as septic system rehabilitation.

No-Till Drill

The no-till drill is an important innovation in tillage practice and is gaining popularity in rural Ontario. Through Land Stewardship II, an Ontario Ministry of Agriculture and Food program, no-till drill demonstrations are underway across the province. Limnological studies comparing rivers in no-till drill soil plots to standard agricultural practice plots showed that the no-till drill conservation technique effectively reduced particulate phosphorus loads (Logan 1982) to the receiving stream. Oloya and Logan (1980) also described the mechanisms and extent to which particulate phosphorus entered lentic systems in rural areas.

In deeper lakes control of dissolved phosphorus may be a higher priority but in shallow reservoirs like Binbrook (where stratification is weak), particulate phosphorus is of greater concern since sediment resuspension by wind and wave action, bioturbation and microfaunal

effects will more readily release this form of P, during the growing season (Syers et al. 1973). Some reduction in particulate phosphorus loads may result from the no-till drill technique.

The Binbrook Wetland

Decreases in concentrations for all key water quality indicator parameters were observed between the inflow and station BR2 just downstream from the wetland area. Continuing sedimentation was observed at BR2 as evidenced by differences in turbidity between surface and bottom waters at this site. This indicates that additional sedimentation upstream of the constructed weir could occur if retention time was increased. Wind-mixing and bottom water draw at station BR1 appear to have had a positive effect preventing the build-up of sedimented particulates at that site, consequently differences in turbidity levels there were negligible.

A wetland's ability to reduce nutrient levels from inflowing waters to reservoirs is well documented (OMOE 1991). Wetlands have been shown to reduce solids levels by 97% and total phosphorus by 50 to 80% (Carlisle and Mulamoottil 1991). The ablities of wetlands to remove nitrogen as well as phosphorus are also well known (Rogers et al. 1991). In a similar situation, but for the control of urban as opposed to agricultural runoff, wetlands are presently being constructed in Kitchener, Ontario and have proven successful in other southern Ontario demonstrations (Herskowitz 1986). The application of the technology to the two situations is similar.

Because the Binbrook Reservoir wetland was not designed for water quality improvements, additional benefits to the reservoir could be expected by enhancing the nutrient removal design specifics, and maintaining the water course through the wetland with additional plantings and substrate alterations for emergent macrophytes. Retrofitting existing wetlands to significantly improve reductions in nutrient and sediment loads downstream, while

maintaining the initial benefits (protection of waterfowl habitat) for which they were principally designed have been previously achieved (Lathrop 1982, OMOE 1991).

From a nutrient control standpoint, the current wetland appears flawed in several areas. Examination of aerial photos shows a wide river down the center of the wetland. A central stream like this allows a good portion of the flow to enter the reservoir directly without wetland residence time. Intercepting the flow and directing it through managed channels into smaller subwetlands could increase plant contact, improve sedimentation and promote microbial activity. OMOE (1991) and Taylor (1992) describe design options which can achieve this goal. Additionally, small islands could be constructed in the middle of the wetland stream to direct flow and sediment laterally (Carlisle and Mulamoottil 1991).

The construction of the weir across the bottom of the reservoir formed a partially effective sedimentation basin which may have initially enhanced the sediment/nutrient containment features of the wetland. The wetland was estimated to contain 485 acre-feet of storage when it was built. It was likely effective in this respect during the first 2-3 years following construction. High sediment loads, however, appear to have quickly displaced its storage capacity. High early season colour levels in the reservoir suggest that the wetland is being flushed prior to the growing season. This process also releases nutrients to the reservoir (Graneli and Solander 1988) and could be elevating reservoir [TP] beyond the contribution from an already high inflow concentration. Nutrients are also contributed to the wetlands by waterfowl. This wetland experiences dense waterfowl populations during the low runoff, autumn period. A large proportion of this phosphorus is expected to remain in the wetland, since plant uptake would not occur in the post-growing season and flushing is low during this time.

Sedimentation basins serving as the primary stage intercepting inflow in front of the wetland, have had positive results by eliminating coarser particles (Lathrop 1982, OMOE 1991), and limiting early season flushing. Evidence from urban stormwater reservoirs which utilized sedimentation basins in their designs, indicated that up to 97% of annual phosphorus and

sediment loads could be contained within the basin (Gemza 1991, OMOE 1991). The predominant soil type in this area, however (clay), is particularly difficult to control by sedimentation because of slow sedimenting velocities. Sedimentation basin effectiveness is reduced with these types of soils, because clay particles are light and have slow settling velocities of approximately 0.012 m/hr (Lathrop 1982). Under these conditions a clay particle would need 100 hours to settle from the top metre of the water column, under calm conditions. Sedimentation basins have been most effective in urban environments where the predominant particle is sand, with settling velocities as high 12 m/hr. As a result, a sedimentation basin alone would have limited success at Binbrook but could serve as an important primary stage in treatment.

Destratification

Complete anoxia or low oxygen concentrations were regularly observed in the deeper waters of the reservoir. Under these conditions the release of sediment bound nutrients and metals are well documented (Syers et al. 1973). Destratification has been shown to reduce or eliminate this release (Lorenzen and Mitchell 1975), maintain fish habitat and improve impoundment and consequently after release, downstream water quality. A variety of techniques have been developed to induce destratification, and selection must be based on a number of governing factors including site layout, electrical power availability, existing equipment, and maintenance considerations. Techniques include aeration using rising columns of bubbles from diffusers mounted on the lake bottoms powered by on-shore air compressors, mechanical devices such as axial flow pumps (Strecker et al. 1977) and jet nozzles inducing a turbulent mixture of air and water through the water column (Stefan and Gu 1991). The latter technique also enhances oxygen transfer into the water directly during application, and during the rising phase of the bubble plume, while the other two techniques rely on oxygen transfer at the water surface only, when direct contact with the atmosphere occurs.

Axial flow pumps are generally more efficient, utilizing motors of lower horsepower to move equivalent volumes of water (Irwin et al. 1969). Combined with high iron concentrations, correlation between iron, phosphorus and turbidity levels, and evidence for the sedimentation of these particles to the lake bottom, prevention of nutrient release by maintaining aerobic conditions at the sediment-water interface is promising. One of the principal reasons for failure of destratification systems in controlling nutrient feedback has been underestimating the sediment ratio of iron:phosphorus, and therefore the phosphorus binding capacity of the sediment. In lakes of low sediment iron, control of phosphorus release by maintaining an oxidized sediment-water interface is less likely to be successful (Jensen et al. 1992).

Zooplankton and the Food Web

The dominance of daphnid species in the zooplankton community of Binbrook reservoir is unusual considering the high proportion of black crappie in the fish community. In recent crappie invasions of southern Georgian Bay (Gemza 1994), Daphnid populations were depressed and seasonal length decreases were observed, as a result of selective predation by the crappie population. As a result, small bodied cladocerans such as *B. longirostris* and predatory cyclopoid invertebrates dominated, and large herbivorous invertebrates were small contributors to the zooplankton population. This is not the case in Binbrook Reservoir where the *Daphnia* retained 50% or more of the population dominance throughout the summer, during which predation by young of the year crappie would have been most intense. In Binbrook reservoir, only large zooplankters like the calanoid *D. siciloides* and to some extent *M. edax* displayed the expected reduction of length and density due to size selective predation pressures.

Tessier (1986) described a dark hypolimnetic refuge reducing predator effectiveness. Diurnal, vertically migrating zooplankters were thus always provided with a constant dark "envelope" where visibility, hence predator success was decreased. A similar effect could be occurring in turbid southern Ontario reservoirs where predator visibility is limited by

Secchi depths less than 1 metre and often closer to 0.5 metres or less. Without visibility the crappie is unable to effectively select for the larger organisms, and relies on random prey encounter. Although a clear statistical evaluation of this proposed phenomenon cannot be made from this limited assessment, minimum zooplankton lengths were observed to occur at the same time as clear water phases were measured (for example, mid-August at station BR1 during 1989).

Food web manipulations which introduced large piscivores into lakes to control zooplanktivores which in turn limited predation on the zooplankton and allowed their numbers to increase resulted in the control of algal levels (Kitchell and Carpenter 1988), because the algae were no longer preyed on by zooplankton. The zooplankton of Binbrook Reservoir however, do not appear stressed by top down effects at this time. Daphnid biomass and percent community dominance exceeding 50%, suggests that food web manipulation at this time would have limited success. Should water clarity increase as a result of wetland enhancement and improved watershed management, zooplankton may become more susceptible to predation and control of the crappie population would then be recommended. As well, a large scale predator transplant involving pike and or walleye, may not be successful at this time because of siltation over spawning beds preventing walleye hatches. Walleye require rocky shoals for spawning which are being covered by clay silt at this time, and are possibly dried out because of fluctuating water levels as a result of winter drawdown, and pike require macrophyte habitat, which is absent from the reservoir. Increased water clarity however, could result in increased macrophyte densities because shading under turbid conditions tends to limit macrophyte growth. This would allow pike an improved chance of success.

The restoration of Binbrook reservoir will require a multi-facetted ecosystem oriented approach:

1) The primary goal should focus on reducing erosion and runoff to the Welland River and improving land management and agricultural practices. This is already underway

through the Binbrook CURB plan.

2) Artificial destratification should reduce sediment nutrient feedback by eliminating anoxic conditions, and enhance fish habitat. Destratification will not, however, reduce turbidity.

3) Enhancing wetland design should improve its nutrient removal and sediment containment features. Combining it with a sedimentation basin should further improve its treatment capabilities. Effectiveness during the spring runoff period might be limited, however, because plant growth is seasonal, and residence time in the basin would be short during periods of high flows, unless sedimentation basin size was designed for this specific control aspect. Good control would be expected during the summer (recreational) season.

4) If water clarity is improved, size selective predation by the crappie population will likely be enhanced, depressing the large bodied zooplankton. Macrophyte densities should also increase. Preparations for increasing the numbers of top predator species should be made to control crappie numbers. A popularized crappie fish harvesting program should be promoted in the interim, if sport fish contaminant levels (currently being examined at MOEE laboratories) are acceptable.

NOTE:

The assessment of reduced nutrient impacts on the trophic state of the reservoir is made difficult because of a lack of flow data for the study period. Calculations of flow based on estimating changes in volume from rule curves of reservoir levels/volume could not be accurately accomplished, consequently the net loadings of nutrients were not determined. There is an absence of any flow measurements even upstream of the reservoir. In order to properly address any design changes to the wetland or sedimentation basin, a detailed knowledge of seasonal flow characteristics is required. With a potential summer seasonal

income of just under \$500,000 based on 70,000 to 80,000 visitors as indicated in Laidley (1988), stream gauging the Welland River as it enters the reservoir should be conducted to quantify success of the CURB in nutrient loading reductions and the future planning of remedial efforts.

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Table 1 Lake physical data and dam specifics.

Normal Reservoir Area	175 hectares
Reservoir area at emergency spillway	227 hectares
Maximum summer depth	8.7 metres
Mean summer depth	3.2 metres
Reservoir length	5.6 km
Average summer width	305 metres
Watershed area	4168 Hectares
Summer storage volume	$5.551 \times 10^6 m^3$ (4,500 acre feet)
Wetland (1981 estimate) storage volume	$5.98 \times 10^5 m^3$ (485 acre feet)
Winter storage volume	$1.26 \times 10^6 m^3$ (1,020 acre feet)
Maximum storage volume at emergency spillway	$6.982 \times 10^6 m^3$ (5,660 acre feet)
Ratio of lake volume : Estimated wetland volume (1981)	9.3 : 1
Controlled outflow rate during summer	141.5 litres/sec 5 cubic feet/sec.

Table 2 Matrices of Pearson, product-moment correlation coefficients related to water clarity (2a), conductivity (2b) and zooplankton (2c). (α=0.01, * not signif.).

2a)	Chlor a total	Fe	Secchi disc	TN:TP ratio	TP	Turbid	SRP
Chlor a total	1.0						
Fe	*0.347	1.0					
Secchi disc	*-.380	-0.585	1.0				
TN:TP	-0.591	-0.545	0.377	1.0			
TP	*0.281	0.767	-0.483	-0.644	1.0		
Turbid	*0.268	0.815	-0.672	-0.603	0.813	1.0	
SRP	*0.157	0.733	-0.480	-0.499	0.885	0.787	1.0

2b)	Fe	Cl	Cond	K	Mg	Mn	Na
Fe	1.0						
Cl	-0.422	1.0					
Cond	-0.434	0.981	1.0				
K	-0.046	0.671	0.642	1.0			
Mg	-0.484	0.652	0.735	*0.213	1.0		
Mn	*0.277	*-.157	*-.147	*0.163	*-.142	1.0	
Na	-0.399	0.874	0.863	0.588	0.541	*-.105	1.0

2c)	Calcop	Cyccop	DC	NDC	TZB	TP	Chlor a total
Calcop	1.0						
Cyccop	*-.104	1.0					
DC	0.628	*-.042	1.0				
NDC	*0.004	0.795	*-.082	1.0			
TZB	*0.278	0.827	*0.299	0.909	1.0		
TP	*0.078	0.595	*-.192	0.632	0.571	1.0	
Chloro	*0.249	*0.262	*-.089	*0.090	0.147	*0.281	1.0

Calanoid copepod (Calcop), Cyclopoid copepod (Cyccop), Daphnid cladoceran (DC), Non-daphnid cladoceran (NDC), Total zooplankton biomass (TZB), Total phosphorus (TP)

Table 3 List of common phytoplankton genera in Binbrook Reservoir, 1987-1989 (* denotes dominant genera).

CYANOPHYTES	**BACILLARIOPHYTES**
Anabaena *	*Asterionella* *
Aphanizomenon *	*Cyclotella* *
Coelosphaerium	*Melosira* *
Lyngbya *	*Nitzschia* *
Oscillatoria	*Stephanodiscus*
	Synedra
DINOPHYTES	
Ceratium	
Peridinium *	
CRYPTOPHYTES	
Cryptomonas *	
Katablepharis	
Rhodomonas	
EUGLENOPHYTES	
Euglena	
Phacus	
Trachelomonas *	
CHRYSOPHYTES	
Chromulina ericensis	
Chrysochromulina parva	
Dinobryon *	
Mallomonas *	
Synura	
CHLOROPHYTES	
Botryococcus *	
Chlamydomonas *	
Closterium	
Gloeocystis *	
Monoraphidium	
Oocystis *	
Pediastrum *	
Quadrigula	
Scenedesmus	
Schroederia	
Staurastrum	

Table 4 List of common limnetic zooplankton in Binbrook Reservoir, 1988-1989.

Daphnid cladocerans
Daphnia parvula
D. galeata mendotae
D. retrocurva
D. pulex
D. ambigua
Non-daphnid cladocerans
Bosmina longirostris
Ceriodaphnia quadrangula
Ceriodaphnia lacustris
Chydorus sphaericus
Diaphanosoma birgei
Eubosmina coregoni
Calanoid copepods
Calanoid copepodid
Diaptomus ashlandi
D. siciloides
D. minutus
D. sanguineus
D. oregonensis
Epischura lacustris
Cyclopoid copepods
Cyclopoid nauplii
Cyclopoid copepodid
Cyclops vernalis
Cyclops bicuspidatus thomasi
Mesocyclops edax
Orthocyclops modestus
Tropocyclops prasinus mexicanus

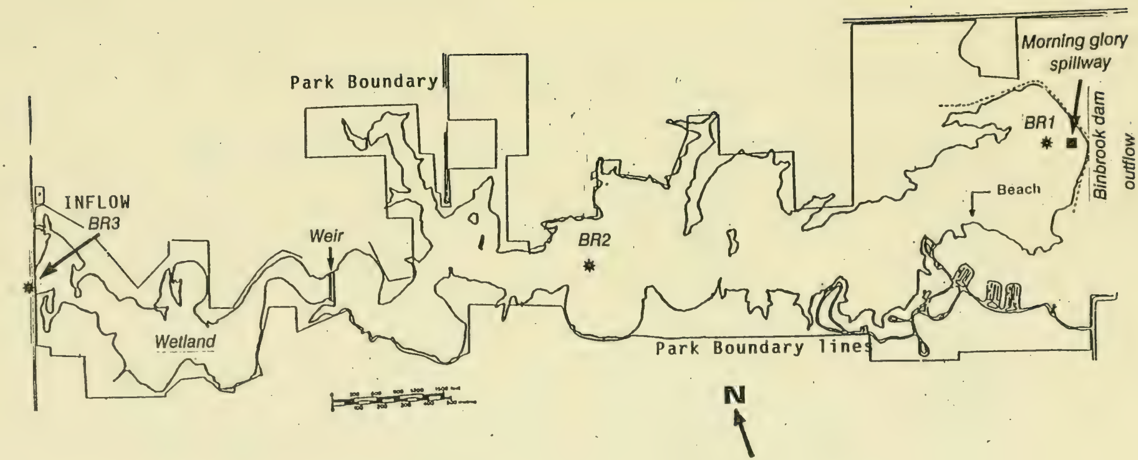
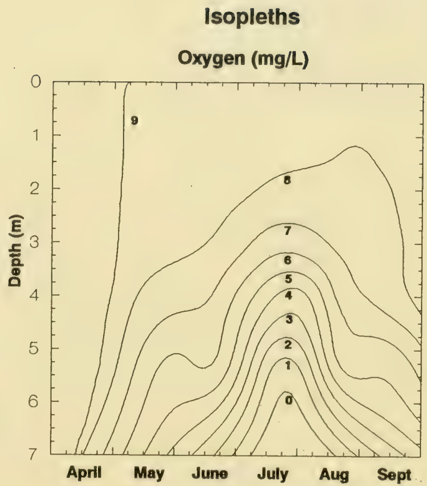


Figure 1: Map of Binbrook Reservoir indicating sampling locations.



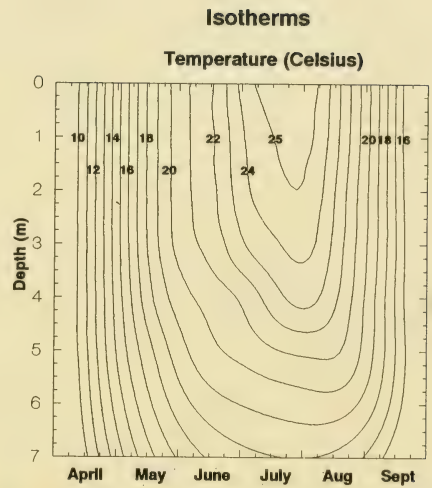


Figure 2: Station BR1, 1988. Oxygen and Temperature profiles from Binbrook Reservoir

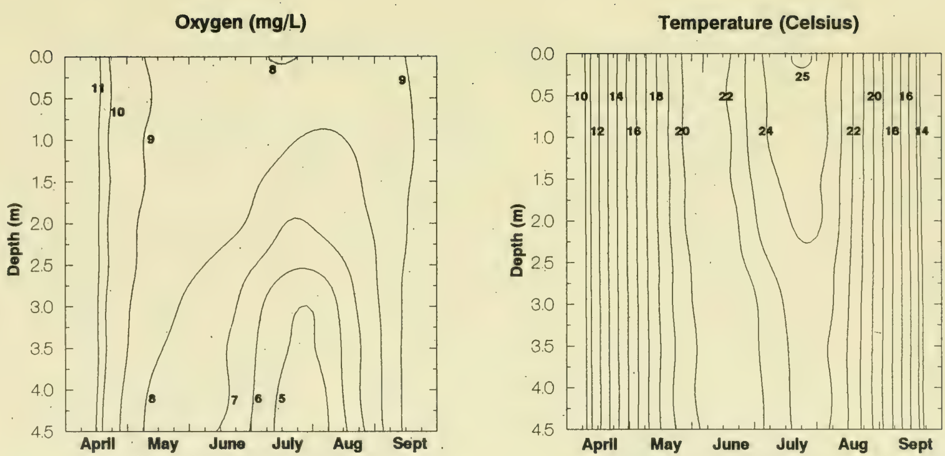


Figure 3: Station BR2 1988. Oxygen and temperature profiles from Binbrook Reservoir.

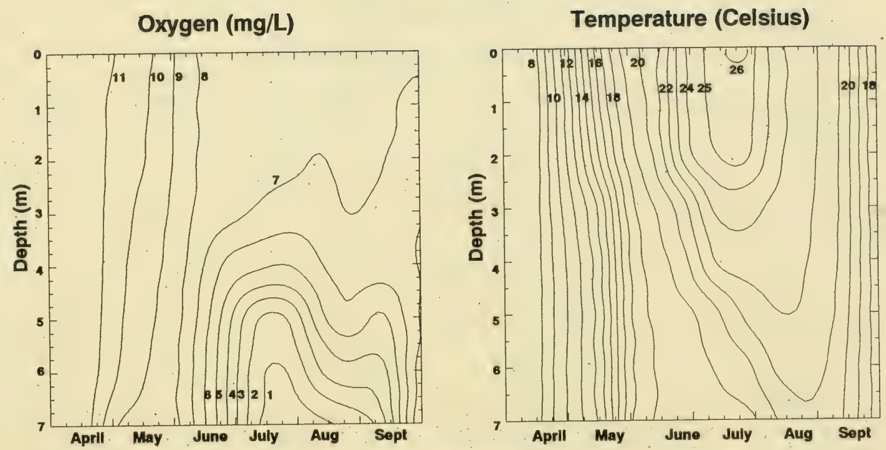


Figure 4: Station BR1, 1989. Oxygen and temperature profiles.

from Binbrook Reservoir.

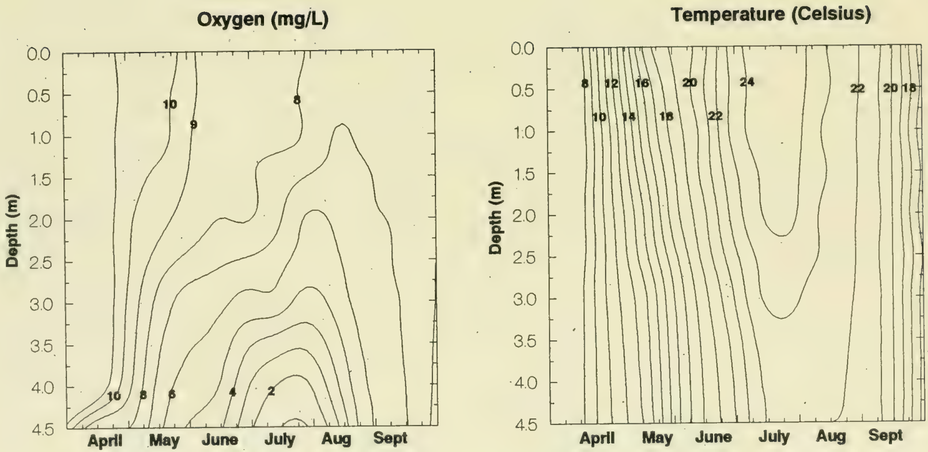


Figure 5: Station BR2 1989, oxygen and temperature profiles from Binbrook Reservoir

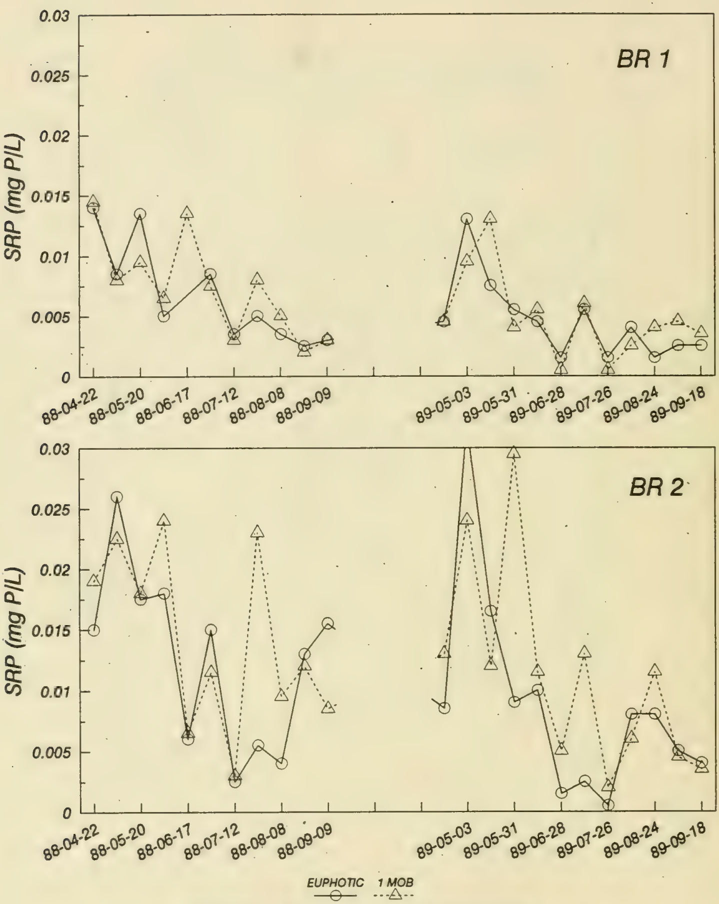


Figure 6: Soluble reactive phosphorus concentration trends during 1988 and 1989.

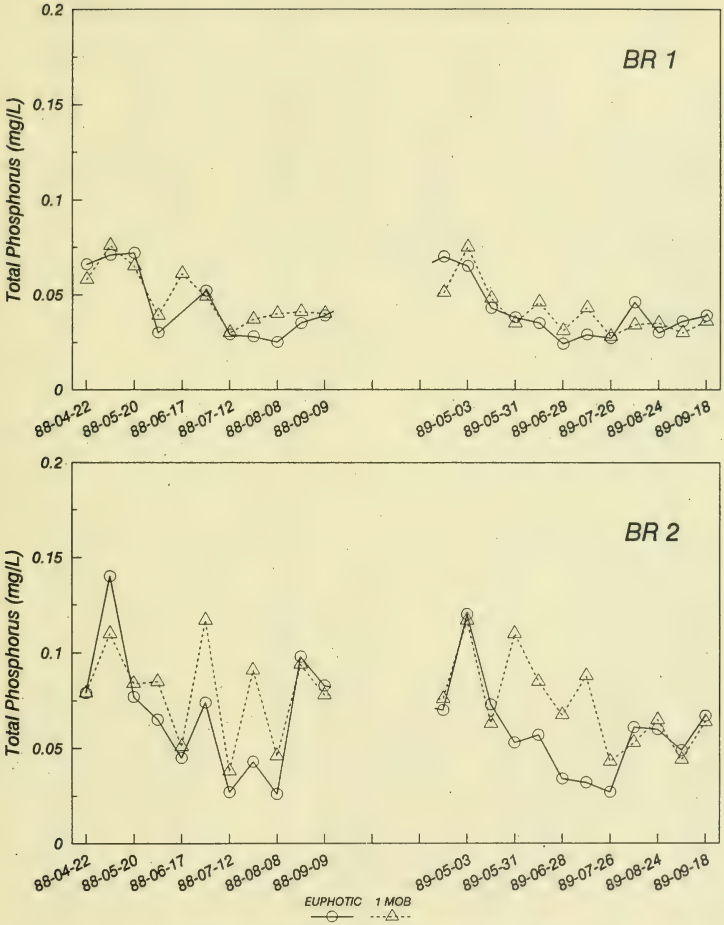


Figure 7: Seasonal total phosphorus trends in Binbrook Reservoir during 1988 and 1989.

Figure 8a: The phosphorus, turbidity relationship in Binbrook Reservoir during 1988 and 1989.

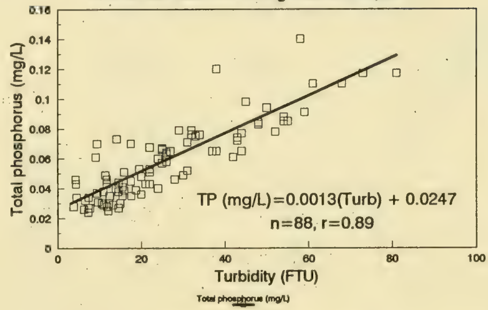


Figure 8b: Secchi disc, chlorophyll a relationship in Binbrook Reservoir during 1988 and 1989.

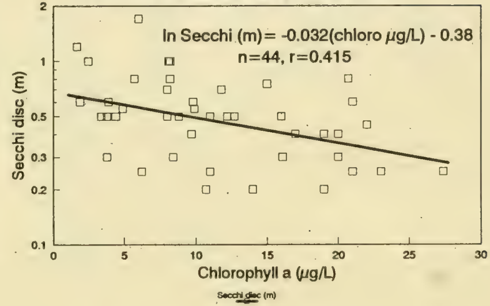


Figure 8c: The Secchi disc, turbidity relationship in Binbrook Reservoir during 1988 and 1989.

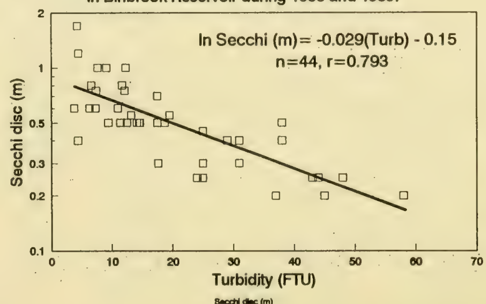
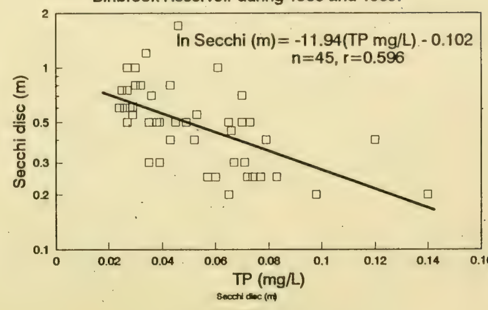


Figure 8d: Secchi disc, TP relationship in Binbrook Reservoir during 1988 and 1989.



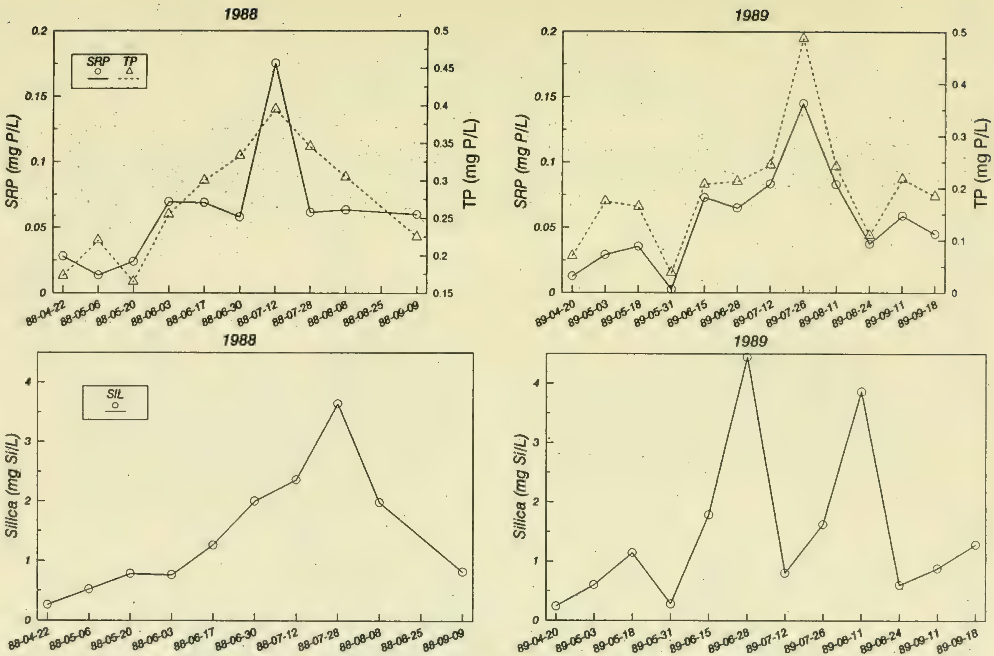


Figure 9: Seasonal phosphorus and silica concentrations of the Welland River inflow to Binbrook Reservoir. 1988 and 1989.

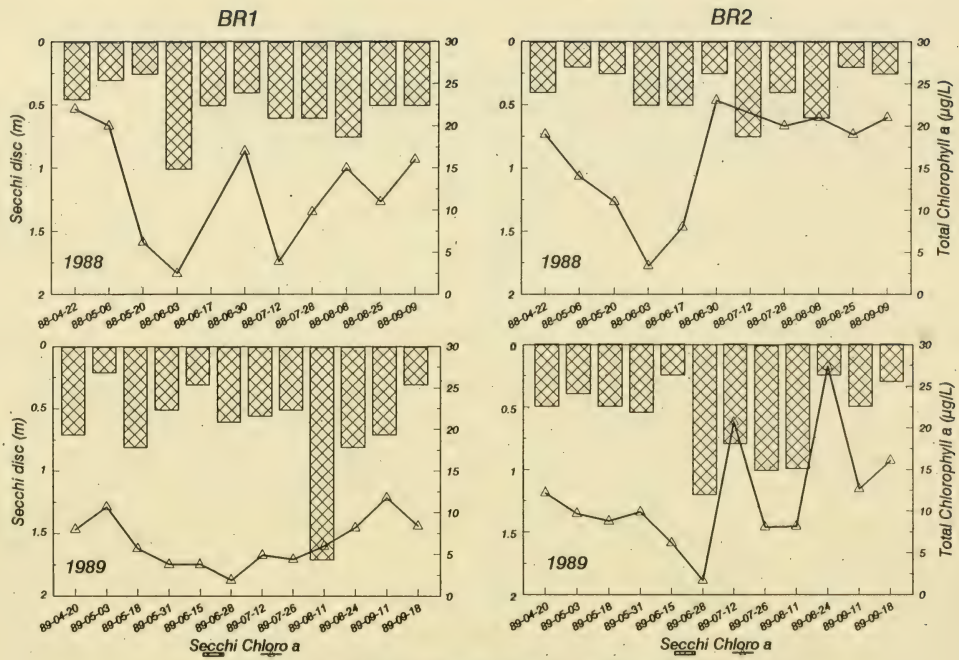


Figure 10: Seasonal secchi disc and total chlorophyll a trends in Binbrook Reservoir. 1988 and 1989.

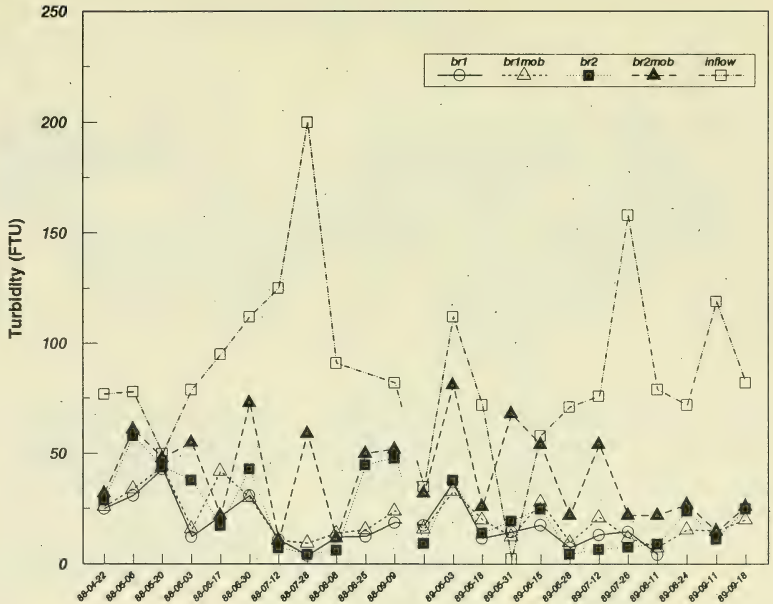


Figure 11: Seasonal turbidity trends in Binbrook Reservoir.

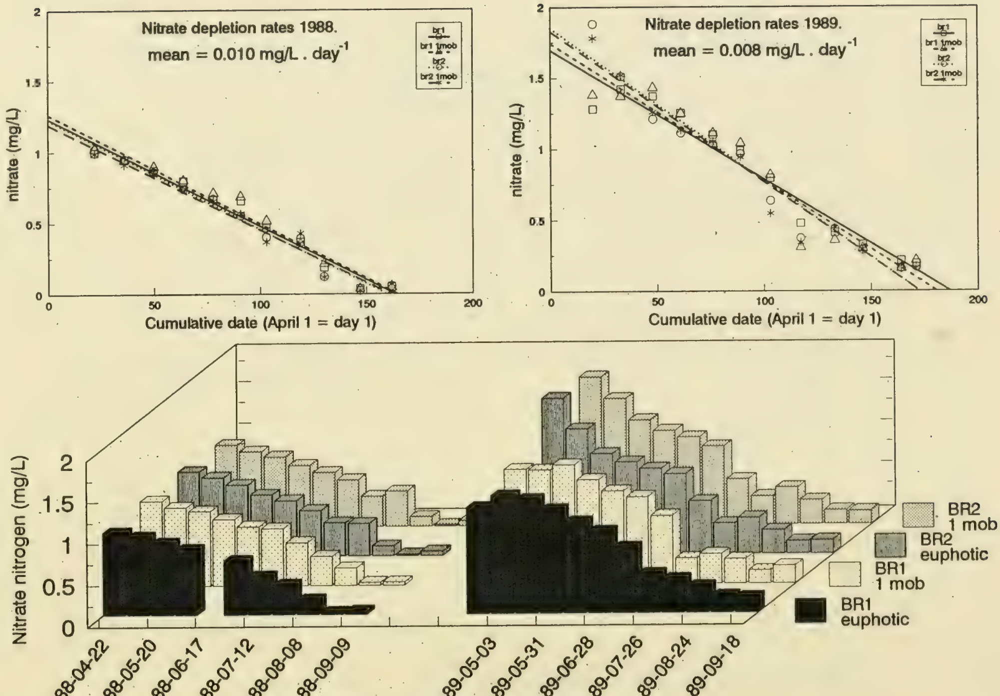


Figure 12: Seasonal nitrate levels in Binbrook reservoir during 1988 and 1989.

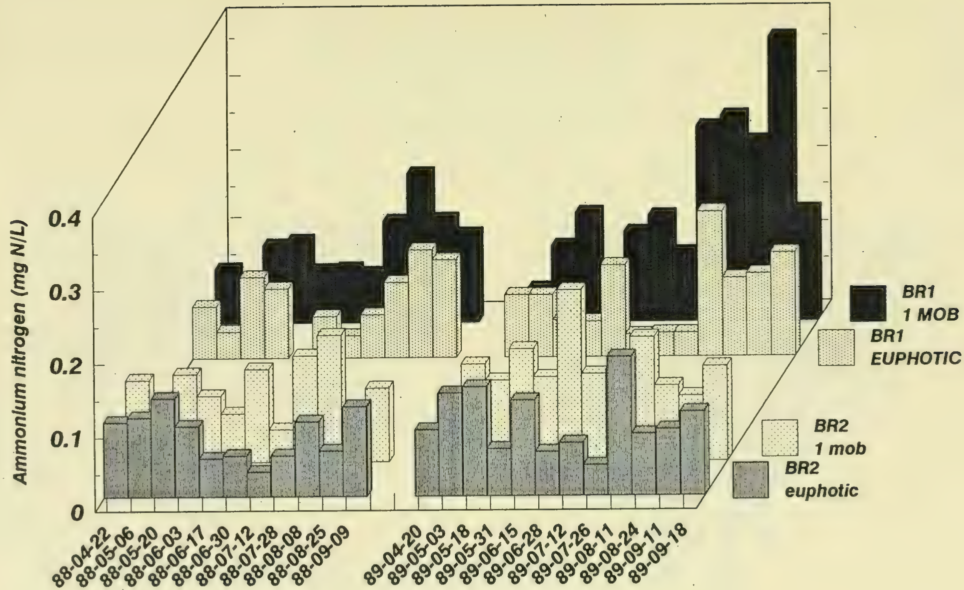


Figure 13: Seasonal ammonium levels in Binbrook

Reservoir during 1988 and 1989.

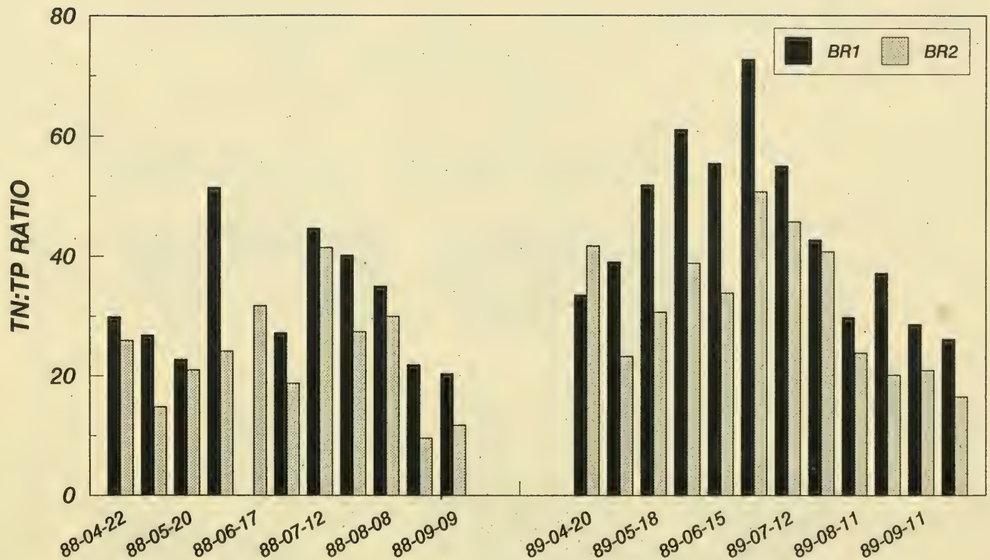


Figure 14: Total nitrogen : Total phosphorus ratios in the euphotic zone at stations BR1 and BR2 during 1988 and 1989.

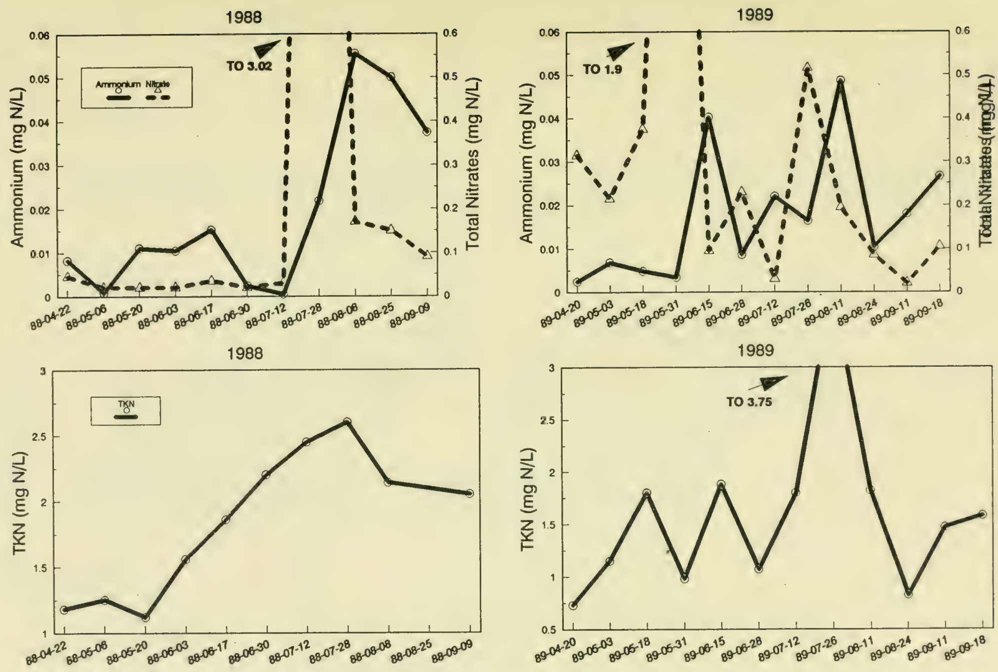


Figure 15: Seasonal nitrogen characteristics of the Welland River inflow at Binbrook Reservoir during 1988 and 1989.

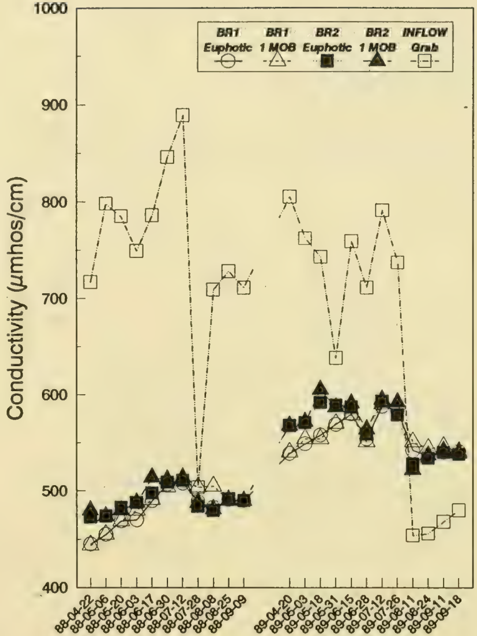


Figure 16a: Seasonal conductivity trends in Binbrook Reservoir.

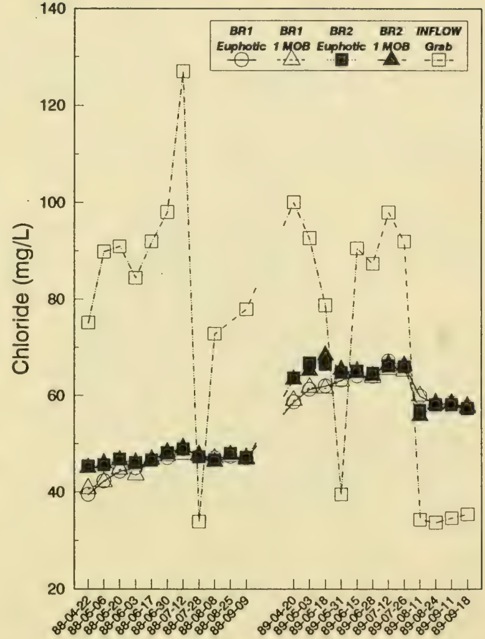


Figure 16b: Seasonal chloride trends in Binbrook Reservoir.

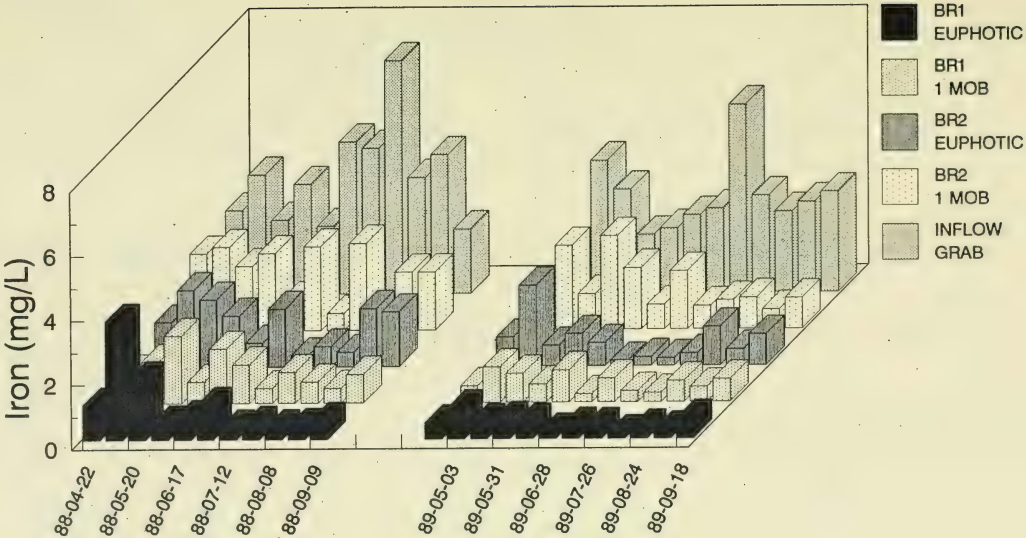


Figure 17: Seasonal iron concentrations in Binbrook Reservoir during 1988 and 1989.

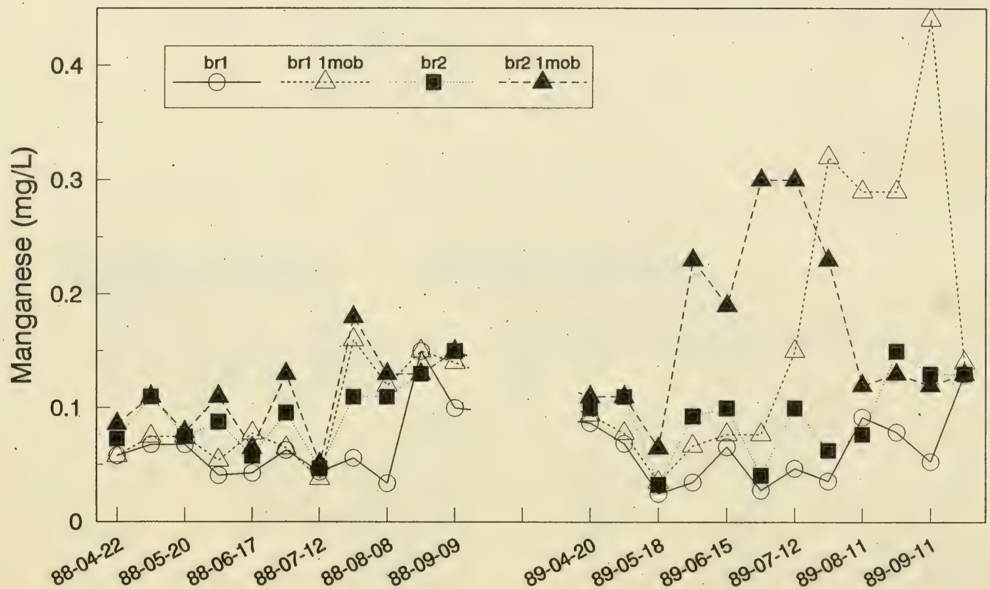


Figure 18: Seasonal manganese concentrations at station BR1 and BR2 during 1988 and 1989.

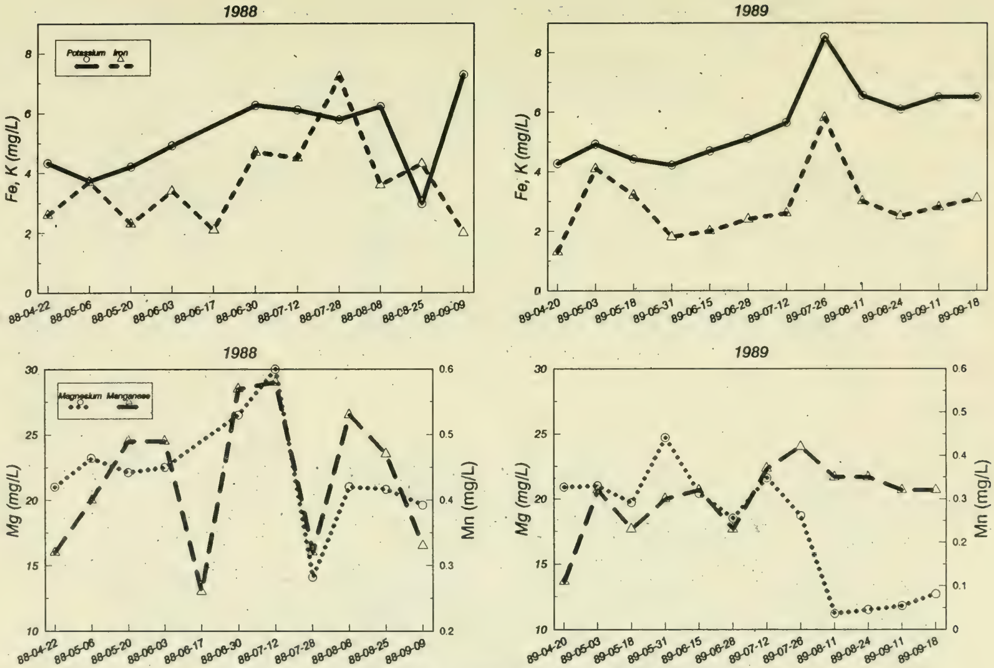


Figure 19: Seasonal metal concentrations in the Welland River inflow to Binbrook Reservoir during 1988 and 1989.

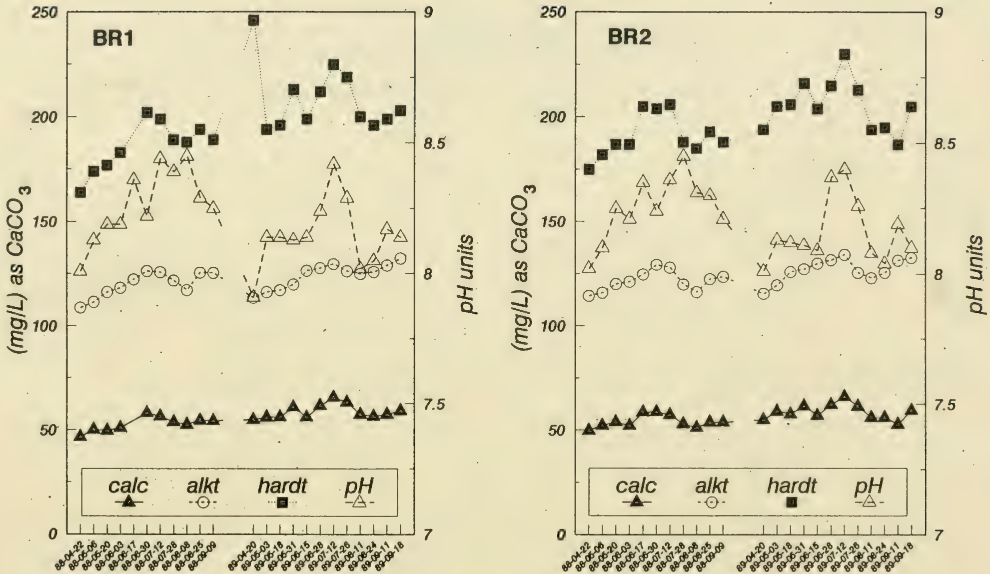
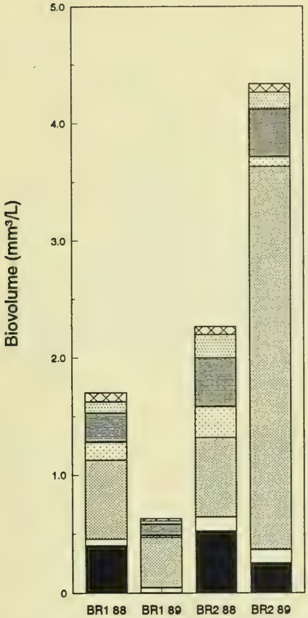
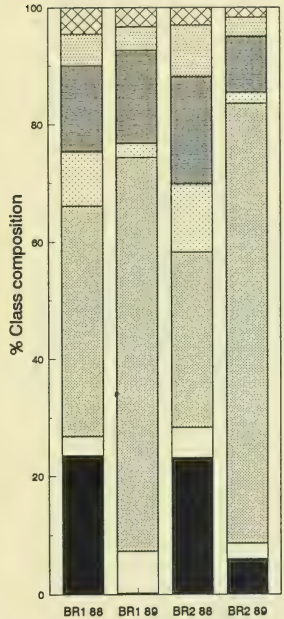


Figure 20: Seasonal euphotic zone calcium, alkalinity, hardness and pH trends from Station's BR1 and BR2 in Binbrook Reservoir during 1988 and 1989.



	BR1 88	BR1 89	BR2 88	BR2 89
CYANO	.400	.001	.525	.253
DINO	.057	.045	.118	.113
CRYPTO	.670	.423	.679	3.267
EUGLENO	.156	.015	.263	.082
CHRYSO	.251	.100	.414	.410
CHLORO	.920	.025	.199	.141
BACILLO	.077	.021	.068	.072



	BR1 88	BR1 89	BR2 88	BR2 89
CYANO	23.6	0.2	23.2	6.0
DINO	3.3	7.1	5.2	2.6
CRYPTO	39.3	67.1	29.9	75.0
EUGLENO	9.2	2.4	11.6	1.9
CHRYSO	14.7	15.9	18.3	9.5
CHLORO	5.4	4.0	8.8	3.3
BACILLO	4.5	3.3	3.0	1.7

Figure 21: Annual recombined phytoplankton biovolume from 1988 and 1989 at Station BR1 and BR2.

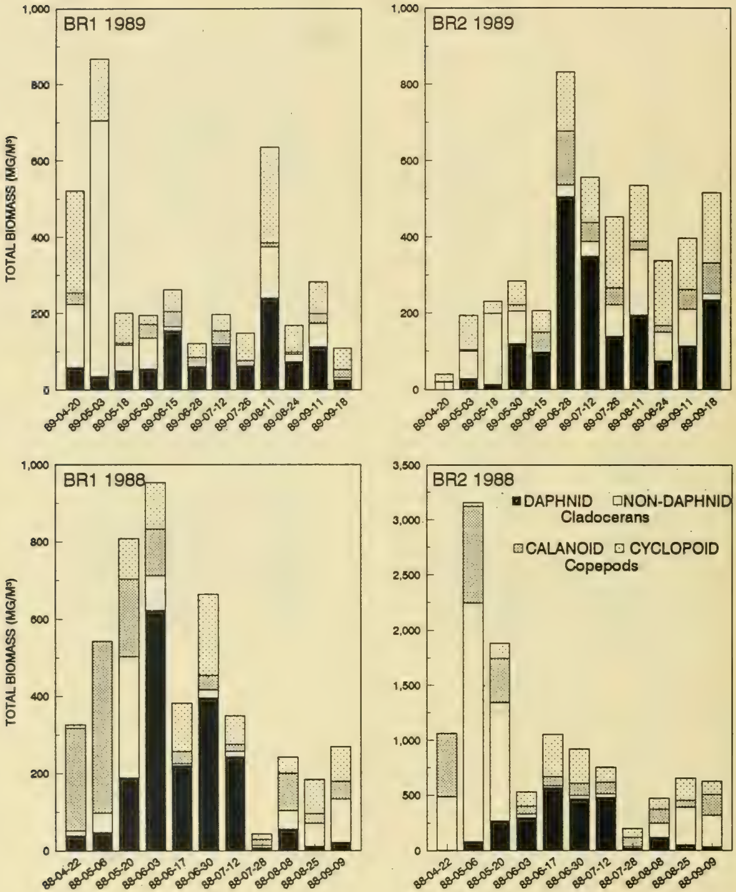


Figure 22: Temporal zooplankton succession at stations BR1 and BR2 during 1988 and 1989.

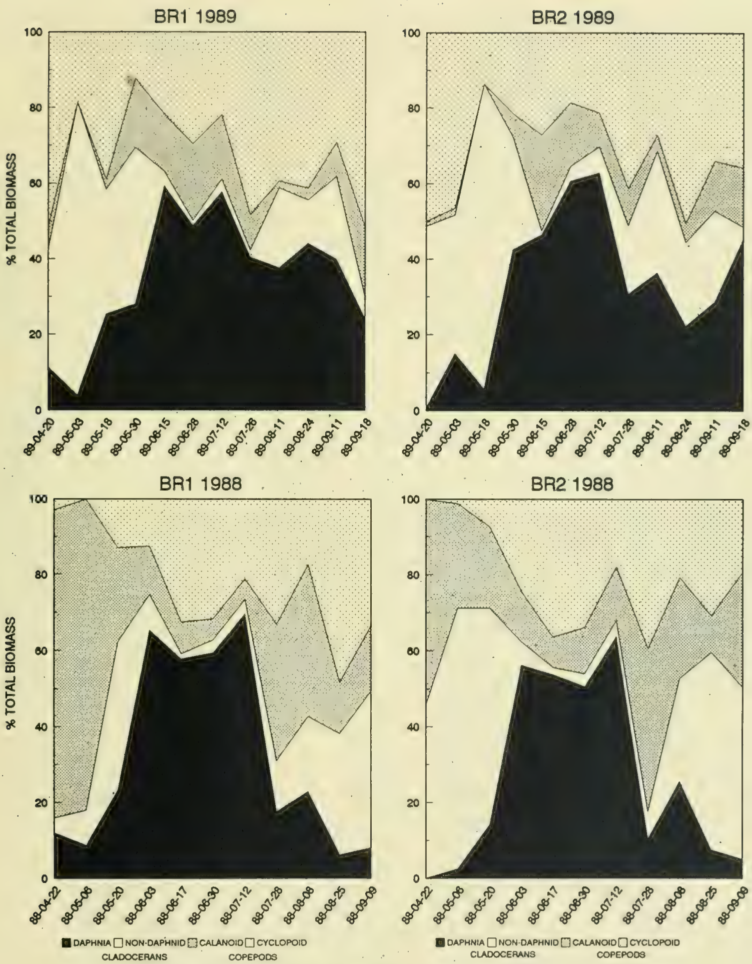


Figure 23: Seasonal zooplankton community structure as a % of total zooplankton biomass. 1988 and 1989.

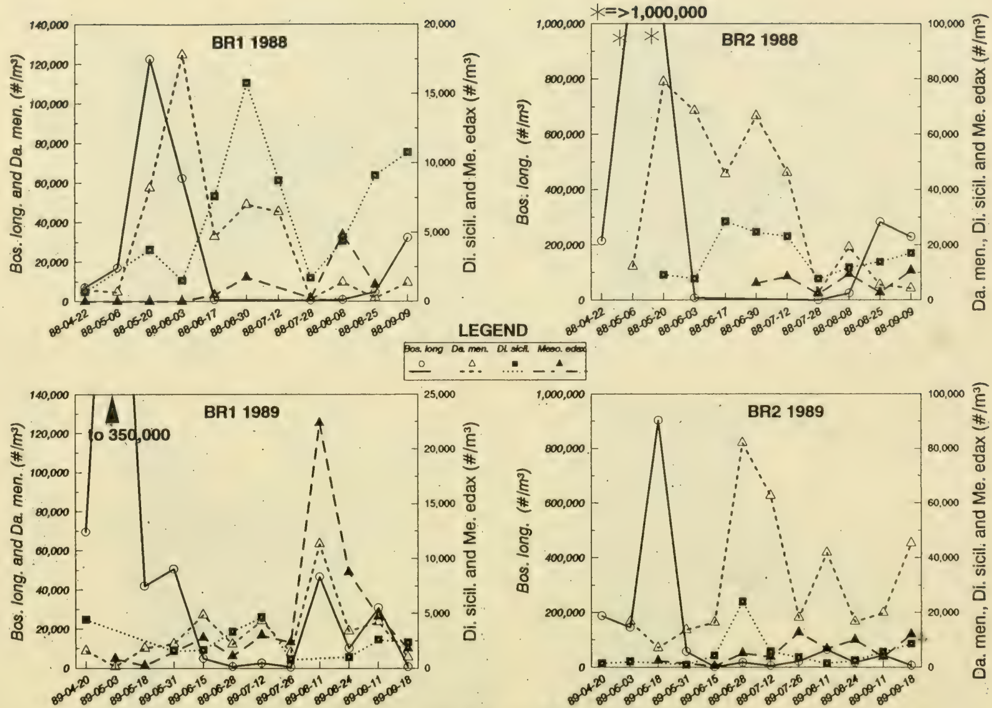


Figure 24: Seasonal density trends of four dominant zooplankton taxa, Bosmina longirostris, Daphnia g. mendotae, Diaptomus siciloides and Mesocyclops edax during 1988 and 1989.

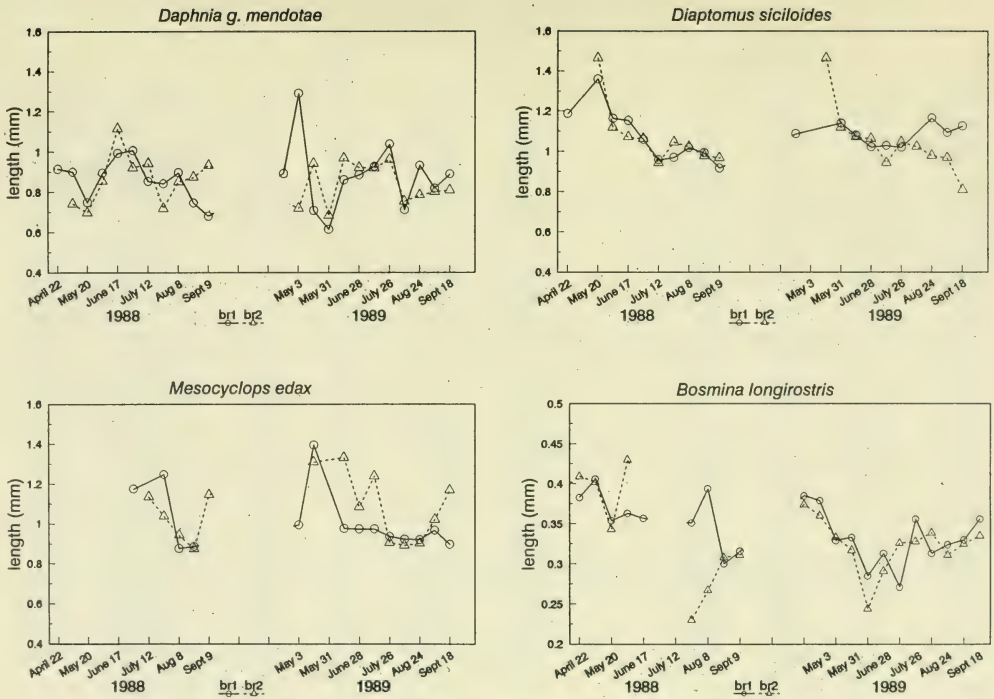


Figure 25: Seasonal length trends of common zooplankton in Binbrook Reservoir during 1988 and 1989.

Appendix A:1988 ice–free season euphotic zone data summary
from station BR1 and BR2.

		1988 Station BR1				1988 Station BR2			
		Mean	STD	Min	Max	Mean	STD	Min	Max
Secchi	(m)	0.532	0.199	0.250	1.000	0.391	0.173	0.200	0.750
Alkalinity	(mg/L)	119.927	5.747	108.800	126.300	121.482	4.607	114.400	129.400
Ca	(mg/L)	52.740	3.285	46.600	58.200	54.118	2.825	49.900	58.800
Chlrac	(µg/L)	10.314	5.680	1.600	18.000	12.344	6.304	2.000	19.000
Chlrat	(µg/L)	12.340	6.389	2.500	22.000	15.940	6.203	3.400	23.000
Chlrbt	(µg/L)	0.957	1.555	0.100	4.700	0.950	0.894	0.100	2.700
Cl	(mg/L)	45.610	2.727	39.500	48.900	46.964	1.028	45.400	48.900
Color	(mg/L)	10.364	4.354	0.500	17.000	12.682	3.892	7.500	20.500
Conduct	(µmho/cm)	481.545	18.995	445.000	508.000	489.273	12.054	473.000	511.000
DIC	(mg/L)	26.620	1.584	24.400	29.000	27.291	1.303	25.600	29.400
DOC	(mg/L)	5.720	0.248	5.500	6.300	5.845	0.246	5.600	6.500
Fe	(mg/L)	1.029	0.949	0.390	3.700	1.362		0.350	2.400
Hard	(mg/L)	185.900	11.086	164.000	202.000	190.909	9.615	175.000	206.000
K	(mg/L)	4.924	0.125	4.720	5.170	4.992	0.144	4.820	5.350
Mg	(mg/L)	13.100	0.837	11.400	14.100	13.482	0.835	12.200	15.300
Mn	(mg/L)	0.066	0.032	0.034	0.150	0.095	0.029	0.047	0.150
Na	(mg/L)	24.780	1.747	21.500	27.100	25.482	0.965	23.900	27.000
NH3	(mg/L)	0.083	0.038	0.030	0.146	0.084	0.031	0.034	0.134
NO3+NO2	(mg/L)	0.536	0.349	0.035	0.995	0.525	0.341	0.020	1.020
TN	(mg/L)	1.334	0.419	0.765	1.975	1.376	0.411	0.780	2.080
TN:TP		32.032	9.965	20.385	51.500	23.353	8.990	9.592	41.481
NO2	(mg/L)	0.025	0.010	0.010	0.047	0.026	0.007	0.011	0.038
TKN	(mg/L)	0.798	0.092	0.690	0.980	0.851	0.133	0.660	1.130
pH		8.265	0.130	8.010	8.450	8.255	0.115	8.020	8.450
SRP	(mg/L)	0.007	0.004	0.003	0.014	0.013	0.007	0.003	0.026
TP	(mg/L)	0.045	0.018	0.025	0.072	0.069	0.032	0.026	0.140
Silica	(mg/L)	0.368	0.163	0.100	0.620	0.418	0.202	0.080	0.780
Sulphate	(mg/L)	58.129	2.990	53.200	62.810	59.166	1.701	56.400	61.380
Turb	FTU	20.040	11.485	3.800	43.000	30.964	18.183	4.400	58.000
Zn	(mg/L)	0.004	0.003	0.001	0.010	0.003	0.002	0.000	0.005

Appendix B:1989 ice–free season euphotic zone data summary from station BR1 and BR2.

		1989 Station BR1				1989 Station BR2			
		Mean	STD	Min	Max	Mean	STD	Min	Max
Secchi	(m)	0.638	0.371	0.200	1.700	0.604	0.306	0.250	1.200
Alkalinity	(mg/L)	124.117	5.717	113.500	132.400	126.925	5.352	115.500	134.200
Ca	(mg/L)	58.808	3.234	54.900	65.700	58.758	3.523	52.900	66.100
Chlrac	(µg/L)	4.533	1.962	2.100	8.700	8.800	5.810	0.800	21.500
Chlrat	(µg/L)	6.467	2.859	1.900	11.800	11.808	6.577	1.700	27.400
Chlrbt	(µg/L)	0.864	0.433	0.400	2.000	1.191	0.563	0.600	2.700
Cl	(mg/L)	61.767	3.095	57.700	67.200	62.833	3.807	56.900	66.800
Color	(mg/L)	13.833	3.399	8.500	19.500	16.417	4.932	10.500	26.500
Conduct	(µmho/cm)	556.667	17.974	537.000	588.000	564.833	23.334	528.000	593.000
DIC	(mg/L)	28.700	1.504	26.400	30.800	29.267	1.292	26.800	31.000
DOC	(mg/L)	6.342	0.225	5.900	6.600	6.625	0.277	6.200	7.200
Fe	(mg/L)	0.473	0.206	0.180	0.970	0.788	0.608	0.160	2.500
Hard	(mg/L)	208.500	14.796	194.000	246.000	205.333	11.462	187.000	230.000
K	(mg/L)	5.357	0.167	5.030	5.570	5.509	0.269	5.150	6.020
Mg	(mg/L)	13.828	0.543	13.200	14.800	14.250	0.842	13.000	15.900
Mn	(mg/L)	0.062	0.030	0.025	0.130	0.094	0.034	0.033	0.150
Na	(mg/L)	33.333	3.173	29.500	42.200	33.925	3.935	29.500	45.400
NH3	(mg/L)	0.087	0.049	0.030	0.196	0.102	0.041	0.042	0.188
NO3+NO2	(mg/L)	0.818	0.453	0.190	1.420	0.813	0.536	0.155	1.880
TN	(mg/L)	1.701	0.542	1.020	2.540	1.752	0.622	1.025	2.920
TN:TP		44.433	14.038	26.154	72.708	32.255	10.838	16.493	50.735
NO2	(mg/L)	0.024	0.004	0.017	0.030	0.029	0.007	0.017	0.042
TKN	(mg/L)	0.883	0.133	0.680	1.120	0.940	0.140	0.730	1.280
pH		8.149	0.125	7.910	8.420	8.158	0.119	8.010	8.400
SRP	(mg/L)	0.005	0.003	0.002	0.013	0.009	0.008	0.001	0.032
TP	(mg/L)	0.040	0.014	0.024	0.070	0.059	0.023	0.027	0.120
Silica	(mg/L)	0.705	0.320	0.200	1.200	0.720	0.374	0.240	1.320
Sulphate	(mg/L)	61.944	3.121	57.000	65.760	60.903	4.318	55.010	66.680
Turb	FTU	15.300	8.721	4.300	37.000	16.192	9.692	4.500	38.000
Zn	(mg/L)	0.002	0.001	0.001	0.004	0.003	0.002	0.001	0.007

Appendix C:1988 ice-free season data summary from
1 meter off bottom samples at station BR1 and BR2.

		1988 Station BR1				1988 Station BR2			
		Mean	STD	Min	Max	Mean	STD	Min	Max
Alkalinity	(mg/L)	122.236	6.985	108.800	130.500	121.991	4.616	116.000	129.600
Ca	(mg/L)	53.791	3.570	46.500	58.600	54.191	2.659	50.900	58.800
Cl	(mg/L)	45.664	2.400	40.700	47.800	46.945	1.124	45.000	49.200
Color	(mg/L)	11.727	3.285	8.500	18.000	12.727	3.701	8.000	18.500
Conduct	(μmho/cm)	485.727	20.671	444.000	510.000	491.909	13.951	474.000	514.000
DIC	(mg/L)	27.600	1.795	24.400	29.600	27.545	1.359	25.800	30.000
DOC	(mg/L)	5.609	0.235	5.300	6.200	5.845	0.284	5.600	6.600
Fe	(mg/L)	1.035	0.489	0.460	2.100	1.851	0.778	0.530	2.700
Hard	(mg/L)	189.000	12.336	163.000	205.000	190.545	8.316	179.000	204.000
pH		8.152	0.105	7.990	8.310	8.202	0.099	7.980	8.340
K	(mg/L)	4.926	0.099	4.770	5.130	5.061	0.182	4.840	5.410
Mg	(mg/L)	13.209	0.902	11.300	14.100	13.345	0.587	12.400	14.100
Mn	(mg/L)	0.092	0.041	0.038	0.160	0.111	0.037	0.051	0.180
Na	(mg/L)	24.873	1.671	21.600	26.900	25.364	0.981	24.100	27.100
NH3	(mg/L)	0.104	0.044	0.036	0.202	0.101	0.037	0.044	0.172
NO3 + NO2	(mg/L)	0.564	0.341	0.035	1.020	0.518	0.330	0.020	0.990
NO2	(mg/L)	0.023	0.011	0.010	0.046	0.025	0.006	0.010	0.036
TKN	(mg/L)	0.800	0.073	0.700	0.960	0.891	0.116	0.740	1.150
TN	(mg/L)	1.364	0.394	0.775	1.980	1.409	0.380	0.890	2.040
TN:TP		28.623	7.229	18.902	41.795	19.053	6.223	9.468	29.079
SRP	(mg/L)	0.007	0.004	0.002	0.015	0.014	0.007	0.003	0.024
TP	(mg/L)	0.049	0.014	0.030	0.076	0.079	0.024	0.038	0.117
Silica	(mg/L)	0.376	0.164	0.060	0.640	0.436	0.203	0.100	0.760
Sulphate	(mg/L)	58.008	2.458	53.200	61.250	59.330	1.954	56.000	62.000
Turbidity	FTU	24.118	11.664	9.400	44.000	43.145	19.888	10.900	73.000
Zn	(mg/L)	0.005	0.003	0.002	0.011	0.006	0.002	0.003	0.010

Appendix D:1989 ice–free season data summary from
1 meter off bottom samples at station BR1 and BR2.

		1989 Station BR1				1989 Station BR2			
		Mean	STD	Min	Max	Mean	STD	Min	Max
Alkalinity	(mg/L)	126.742	7.870	111.900	139.000	128.033	5.972	115.400	136.400
Ca	(mg/L)	58.725	3.946	53.400	66.800	59.467	4.000	54.100	67.200
Cl	(mg/L)	61.592	2.727	57.800	65.400	62.800	4.027	55.800	68.300
Color	(mg/L)	14.375	2.686	11.500	20.500	16.083	4.266	11.500	24.000
Conduct	(μmho/cm)	559.167	17.559	540.000	590.000	567.833	26.124	522.000	605.000
DIC	(mg/L)	29.550	2.160	26.200	33.600	29.933	1.571	26.800	31.800
DOC	(mg/L)	6.550	0.574	6.000	8.300	6.658	0.250	6.300	7.100
Fe	(mg/L)	0.629	0.263	0.250	1.100	1.358	0.731	0.600	2.900
Hard	(mg/L)	203.750	11.924	189.000	228.000	207.167	13.366	190.000	234.000
pH		8.083	0.126	7.770	8.260	8.082	0.114	7.930	8.310
K	(mg/L)	5.453	0.206	5.160	5.920	5.693	0.447	5.240	7.000
Mg	(mg/L)	13.892	0.598	13.300	14.900	14.258	0.911	12.900	16.100
Mn	(mg/L)	0.172	0.124	0.035	0.440	0.170	0.075	0.065	0.300
Na	(mg/L)	33.417	2.888	31.200	42.400	34.017	3.873	28.700	44.800
NH3	(mg/L)	0.170	0.097	0.046	0.384	0.130	0.038	0.088	0.232
NO3 + NO2	(mg/L)	0.810	0.490	0.155	1.430	0.803	0.528	0.160	1.780
NO2	(mg/L)	0.028	0.007	0.021	0.046	0.033	0.011	0.015	0.057
TKN	(mg/L)	1.005	0.111	0.890	1.330	1.045	0.179	0.850	1.400
TN	(mg/L)	1.815	0.529	1.190	2.700	1.848	0.650	1.020	2.910
TN:TP		44.984	9.590	33.056	64.194	25.614	6.210	17.813	38.289
SRP	(mg/L)	0.005	0.003	0.001	0.013	0.011	0.008	0.002	0.030
TP	(mg/L)	0.041	0.012	0.028	0.075	0.073	0.023	0.043	0.117
Silica	(mg/L)	0.783	0.274	0.300	1.200	0.750	0.371	0.220	1.320
Sulphate	(mg/L)	59.596	3.932	54.340	65.400	61.048	4.712	54.520	68.600
Turbidity	FTU	17.475	7.126	7.400	33.000	37.383	20.447	14.600	81.000
Zn	(mg/L)	0.009	0.005	0.006	0.023	0.009	0.004	0.004	0.016

Appendix E: Ice free season data summary from grab samples collected from the Welland River inflow to Binbrook Reservoir during 1988 and 1989.

		BR3 1988				BR3 1989			
		Mean	STD	Min	Max	Mean	STD	Min	Max
Alkalinity	(mg/L)	193.173	33.812	109.500	231.800	175.217	27.859	135.200	216.200
Ca	(mg/L)	77.940	9.709	57.400	91.200	72.058	11.189	54.900	87.100
Cl	(mg/L)	84.170	22.227	33.900	127.000	68.033	27.974	33.700	100.000
Color	(mg/L)	26.750	6.619	19.000	43.500	30.583	12.074	0.500	49.500
Conduct	(μmho/cm)	747.455	94.668	504.000	889.000	650.333	137.383	454.000	805.000
DIC	(mg/L)	43.280	7.591	24.800	52.000	39.950	5.687	31.200	48.000
DOC	(mg/L)	10.870	2.393	7.100	15.000	9.117	1.831	6.000	12.200
Fe	(mg/L)	3.673	1.445	2.000	7.200	2.883	1.121	1.300	5.800
Hard	(mg/L)	285.900	39.622	202.000	349.000	252.917	44.599	186.000	306.000
pH		8.182	0.153	7,840	8.410	8.134	0.133	7.920	8.380
K	(mg/L)	5.179	1.284	2.980	7.270	5.618	1.219	4.220	8.510
Mg	(mg/L)	22.080	3.969	14.100	30.000	17.725	4.461	11.200	24.700
Mn	(mg/L)	0.433	0.107	0.260	0.580	0.303	0.077	0.110	0.420
Na	(mg/L)	45.500	10.404	22.300	63.500	37.233	15.691	18.600	66.500
NH3	(mg/L)	0.163	0.168	0.004	0.554	0.174	0.142	0.024	0.486
NO3 + NO2	(mg/L)	0.347	0.892	0.020	3.020	0.340	0.491	0.020	1.900
NO2	(mg/L)	0.043	0.057	0.006	0.199	0.032	0.011	0.012	0.059
TKN	(mg/L)	1.841	0.509	1.120	2.600	1.571	0.767	0.730	3.750
TN	(mg/L)	2.188	1.233	1.140	5.620	1.911	0.877	0.910	4.265
TN:TP		7.860	2.980	5.773	16.290	14.672	18.587	6.103	75.789
SRP	(mg/L)	0.062	0.042	0.014	0.175	0.056	0.037	0.003	0.145
TP	(mg/L)	0.272	0.072	0.165	0.395	0.196	0.108	0.038	0.487
Silica	(mg/L)	1.438	0.992	0.260	3.640	1.460	1.292	0.240	4.440
Sulphate	(mg/L)	82.606	6.975	70.900	93.000	60.666	22.469	42.120	116.620
Turbidity	FTU	98.900	38.911	50.000	200.000	78.033	38.017	2.400	158.000
Zn	(mg/L)	0.010	0.005	0.000	0.018	0.011	0.004	0.006	0.021

Appendix F: Selected key raw water quality data from the euphotic zones of Station BR1 and BR2, and the Welland River inflow collected during the study period.

******************************STATION BR1 MAIN******************************

DATE	SECCHI	CHLRAT	CI	CONDUCT	DIC	Fe	Si	Mn	NH3	NO3+NO2	NO2	SRP	TP	TURBIDITY
88-04-22	0.45	22	39.5	445	24.4	1.1	0.6	0.058	0.07	0.995	0.019	0.014	0.066	25
88-05-06	0.3	20	42.3	455	25.2	3.7	0.44	0.068	0.036	0.945	0.016	0.0065	0.071	31
88-05-20	0.25	6.2	44.3	469	25.6	1.9	0.44	0.068	0.11	0.87	0.028	0.0135	0.072	43
88-06-03	1	2.5	45	470	27.2	0.52	0.3	0.041	0.094	0.795	0.019	0.005	0.03	12.3
88-06-17	0.5			490		0.69		0.043						
88-06-30	0.4	17	47.1	507	25	1.1	0.14	0.063	0.056	0.655	0.047	0.0085	0.052	31
88-07-12	0.6	3.9	48.9	508	29	0.39	0.1	0.044	0.03	0.465	0.031	0.0035	0.029	11
88-07-28	0.6	9.8	47.5	489	27.2	0.5	0.32	0.056	0.058	0.365	0.032	0.005	0.028	3.8
88-08-08	0.75	15	47.1	483	25.6	0.39	0.3	0.034	0.102	0.185	0.021	0.0035	0.025	12.1
88-08-25	0.5	11	47.4	491	28.6	0.43	0.42	0.15	0.146	0.035	0.01	0.0025	0.035	12.5
88-09-09	0.5	16	47	490	28.4	0.6	0.62	0.1	0.132	0.045	0.022	0.003	0.039	18.7
89-04-20	0.7	8	58.7	539	26.4	0.51	1.2	0.087	0.084	1.28	0.021	0.0045	0.07	17.5
89-05-03	0.2	10.7	61.3	549	26.6	0.97	1.06	0.069	0.084	1.42	0.025	0.013	0.065	37
89-05-18	0.8	5.7	62	558	26.6	0.55	1.04	0.025	0.05	1.37	0.024	0.0075	0.043	11.7
89-05-31	0.5	3.8	63.2	569	27.2	0.61	0.9	0.035	0.048	1.25	0.025	0.0055	0.038	14.5
89-06-15	0.3	3.8	64.1	580	29.6	0.55	0.92	0.066	0.124	1.1	0.03	0.0045	0.035	17.6
89-06-28	0.6	1.9	64	554	29.2	0.24	0.84	0.028	0.03	0.985	0.03	0.0015	0.024	7.3
89-07-12	0.55	4.9	67.2	588	30	0.41	0.66	0.047	0.032	0.795	0.024	0.0055	0.029	13.2
89-07-26	0.5	4.4	66.3	583	29.2	0.4	0.56	0.036	0.032	0.475	0.023	0.0015	0.027	14.6
89-08-11	1.7	6	59.9	542	28.8	0.18	0.36	0.092	0.196	0.41	0.027	0.004	0.046	4.3
89-08-24	0.8	8.2	58.3	537	30	0.34	0.42	0.079	0.106	0.325	0.024	0.0015	0.03	
89-09-11	0.7	11.8	58.5	540	30.8	0.28	0.2	0.053	0.112	0.21	0.017	0.0025	0.036	
89-09-18	0.3	8.4	57.7	541	30	0.64	0.3	0.13	0.14	0.19	0.021	0.0025	0.039	

**********************************STATION BR2********************************

DATE	SECCHI	CHLRAT	CI	CONDUCT	DIC	Fe	Si	Mn	NH3	NO3+NO2	NO2	SRP	TP	TURBIDITY
88-04-22	0.4	19	45.4	473	25.6	1.4	0.56	0.073	0.102	1.02	0.02	0.015	0.079	29
88-05-06	0.2	14	45.6	474	26	2.4	0.48	0.11	0.108	0.95	0.023	0.026	0.14	58
88-05-20	0.25	11	46.8	483	26	2.1	0.46	0.075	0.134	0.86	0.029	0.0175	0.077	44
88-06-03	0.5	3.4	46.1	488	27.8	1.6	0.36	0.088	0.096	0.74	0.024	0.018	0.065	38
88-06-17	0.5	8	46.6	498	28.2	0.75	0.4	0.058	0.052	0.67	0.026	0.006	0.045	17.5
88-06-30	0.25	23	48.1	509	29.4	1.8	0.08	0.096	0.056	0.55	0.038	0.015	0.074	43
88-07-12	0.75		48.9	511	29	0.35	0.08	0.047	0.034	0.4	0.029	0.0025	0.027	7.4
88-07-28	0.4	20	47.3	484	26.4	0.63	0.48	0.11	0.056	0.39	0.031	0.0055	0.043	4.4
88-08-08	0.6	21	46.6	480	25.8	0.45	0.3	0.11	0.102	0.12	0.019	0.004	0.026	6.3
88-08-25	0.2	19	48	492	28	1.8	0.62	0.13	0.062	0.02	0.011	0.013	0.098	45
88-09-09	0.25	21	47.2	490	28	1.7	0.78	0.15	0.122	0.055	0.032	0.0155	0.063	48
89-04-20	0.5	12.2	63.6	568	26.8	0.9	1.32	0.1	0.09	1.88	0.029	0.0085	0.07	9.4
89-05-03	0.4	9.7	66.8	571	27.4	2.5	1.08	0.11	0.14	1.51	0.035	0.032	0.12	38
89-05-18	0.5	8.8	66.5	592	28.6	0.64	1.16	0.033	0.148	1.21	0.026	0.0165	0.073	14.1
89-05-31	0.55	9.9	64.8	589	29	1	1.06	0.093	0.064	1.11	0.031	0.009	0.053	19.5
89-06-15	0.25	6.2	65.1	587	30.2	0.71	0.98	0.1	0.13	1.03	0.042	0.01	0.057	25
89-06-28	1.2	1.7	64.6	559	30.4	0.16	0.84	0.041	0.08	0.965	0.031	0.0015	0.034	4.5
89-07-12	0.8	20.7	66.2	593	30.8	0.27	0.5	0.1	0.072	0.635	0.027	0.0025	0.032	6.6
89-07-26	1	8.1	66	579	28.8	0.23	0.48	0.063	0.042	0.37	0.024	0.0005	0.027	7.7
89-08-11	1	8.2	56.9	528	28.2	0.38	0.34	0.077	0.188	0.435	0.034	0.006	0.061	9.1
89-08-24	0.25	27.4	58.1	534	30	1.2	0.3	0.15	0.084	0.285	0.029	0.008	0.06	24
89-09-11	0.5	12.7	58.1	540	31	0.5	0.24	0.13	0.09	0.155	0.017	0.005	0.049	11.4
89-09-18	0.3	16.1	57.3	538	30	0.96	0.34	0.13	0.114	0.165	0.019	0.004	0.067	25

****************************STATION BR3 WELLAND RIVER TYNESIDE RD**********

DATE	CI	CONDUCT	DIC	Fe	Si	Mn	NH3	NO3+NO2	NO2	SRP	TP	TURBIDITY
88-04-22	75.1	717	44	2.6	0.26	0.32	0.082	0.045	0.011	0.028	0.173	77
88-05-06	89.8	798	48	3.7	0.52	0.4	0.008	0.02	0.006	0.0135	0.22	78
88-05-20	90.9	785	44.6	2.3	0.78	0.49	0.11	0.02	0.007	0.024	0.165	50
88-06-03	84.4	749	44	3.4	0.76	0.49	0.104	0.02	0.018	0.0695	0.255	79
88-06-17	91.9	786	48	2.1	1.26	0.26	0.152	0.035	0.019	0.069	0.3	95
88-06-30	98	846	51	4.7	2	0.57	0.024	0.02	0.017	0.058	0.333	112
88-07-12	127	889	52	4.5	2.36	0.58	0.004	0.03	0.025	0.175	0.395	125
88-07-28	33.9	504	24.8	7.2	3.64	0.32	0.218	3.02	0.199	0.0615	0.345	200
88-08-08	72.8	709	38	3.6	1.98	0.53	0.554	0.17	0.086	0.0635	0.305	91
88-08-25		728		4.3		0.47						
88-09-09	77.9	711	38.4	2	0.82	0.33	0.374	0.09	0.037	0.06	0.225	82
89-04-20	100	805	39.6	1.3	0.24	0.11	0.024	0.315	0.012	0.0125	0.07	35
89-05-03	92.6	782	39	4.1	0.6	0.32	0.068	0.215	0.023	0.029	0.175	112
89-05-18	78.7	743	45.8	3.2	1.14	0.23	0.048	0.375	0.044	0.0355	0.165	72
89-05-31	39.5	638	34.4	1.8	0.28	0.3	0.034	1.9	0.039	0.0025	0.038	2.4
89-06-15	90.5	759	46	2	1.78	0.32	0.402	0.095	0.031	0.073	0.208	58
89-06-28	87.3	711	46	2.4	4.44	0.23	0.086	0.23	0.036	0.065	0.213	71
89-07-12	97.9	791	48	2.6	0.8	0.37	0.22	0.03	0.022	0.0835	0.245	76
89-07-26	91.9	737	45	5.8	1.62	0.42	0.164	0.515	0.033	0.145	0.487	158
89-08-11	34.3	454	31.2	3	3.86	0.35	0.486	0.195	0.059	0.083	0.242	79
89-08-24	33.7	456	33.6	2.5	0.6	0.35	0.104	0.085	0.029	0.0375	0.11	72
89-09-11	34.6	468	35	2.8	0.88	0.32	0.18	0.02	0.027	0.059	0.218	119
89-09-18	35.4	480	35.8	3.1	1.28	0.32	0.266	0.105	0.034	0.045	0.185	82